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# Evaluating Alternative Technologies And Monitoring Methods For Water Quality In A Field Setting; Research On Effects On Phosphorous And Solids Removal From Cheese Factory Wash Water And Stormwater Runoff Treatment

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EVALUATING ALTERNATIVE TECHNOLOGIES AND MONITORING METHODS  
FOR WATER QUALITY IN A FIELD SETTING; RESEARCH ON EFFECTS ON  
PHOSPHOROUS AND SOLIDS REMOVAL FROM CHEESE FACTORY WASH  
WATER AND STORMWATER RUNOFF TREATMENT.

A Thesis Presented

by

Dana Allen

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The Faculty of the Graduate College

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for the Degree of Master of Science  
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## **ABSTRACT**

Lake Champlain is a major economic driver for Vermont's tourism economy, as well as a primary source of drinking water for many of the state's residents but nutrient pollution represents a potential threat to ecosystem health and economic well-being. From December 2011 to December 2012 a field trial of an EAF steel slag filter was assessed for its feasibility in treating wastewater originating from Swan Valley Cheese (SVC), in Swanton, VT. The study focuses on a period of the filter's operation from May 4 to October 10, 2012. The plant generates approximately 20,000 gallons per day of high P concentration wash water which is treated in an open aerated lagoon. The filter treated effluent from this lagoon. The major goals of this research were to conduct a field trial of an EAF steel slag filter to evaluate its effect on total P (TP), dissolved reactive P (DRP), and total suspended solids (TSS). Research was also conducted on pH reduction for filter effluent.

Results indicate that the filter removed 95.83% of TP, 96.65% of DRP, and 52.25% of TSS. Average pH effluent was measured at  $10.12 \pm 1.55$ .

Additionally, a field study was conducted on sampling two unlined bioretention systems treating urban stormwater runoff. Methods used are presented and methodological considerations for future studies are presented to guide researchers in more effective and efficient methods for obtaining influent and effluent samples from bioretention systems that are not necessarily designed for sampling.

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## **1 LITERATURE REVIEW**

### ***1.1 Eutrophication and Cyanobacteria***

Nutrient loading from human activities causing eutrophication in waterways has been recognized as a leading cause of water quality impairment for over 50 years (Conley et al., 2009; Ryther and Dunstan, 1971; Schelske, 2009; Schindler, 1974). Research conducted by the World Resources Institute reported over 415 areas globally suffering from eutrophication, demonstrating the widespread, global nature of the problem. In the United State and Europe, recent coastal surveys indicate that 78% of the U.S. coast and 65% of the European coast show some signs of eutrophication linked to nutrient pollution (Selman, et al., 2013). Severe eutrophication resulting in hypoxia, where dissolved oxygen levels in water drop below the threshold at which most aquatic life can survive, has exhibited exponential growth over the past 50 years; between 1960 and 2007 incidences of hypoxia increased 20 fold from 10 to 169 areas (Selman et al., 2013) Despite this increase in the knowledge and awareness about eutrophication, the actual extent of eutrophication is likely underrepresented as many areas of the world are just beginning to look at the problem. Asia, despite its exponential growth in population and industrial development, is the most underrepresented region in terms of eutrophication identification, followed by Latin America, the Caribbean, and Africa (Selman et al., 2013).

Nitrogen (N) is generally the limiting nutrient in estuarine and marine aquatic ecosystems while Phosphorus (P) typically limits algal growth in freshwater lakes, rivers, and streams (Gold and Sims, 2005; Ryther and Dunstan, 1971; Schindler et al., 2008; Smith, 1982). It has been shown that 1g of P as phosphate can cause up to 100g of algae growth (Putz,

2008). Much of this pollution is due to changes in agricultural practices, specifically the application of fertilizers containing high levels of N and P, as well as potassium (Kleinman et al., 2011; Matson, et al., 1997; McDowell & Sharpley, 2001; Tilman, 1999). In freshwater ecosystems excess P in the system can result in a bloom of primary photosynthetic producers in the cyanobacteria family, commonly known as blue-green algae (Gold & Sims, 2005; Smith, 1982; Smith, et al., 1999; USGS, 1999). Blooms can be significant enough to prevent human recreational activities such as swimming, fishing, and boating through the density of the bloom or the often unpleasant odor which accompanies the growth (Fish et al., 2002; Smith, 2003; Smith and Schindler, 2009; USGS, 1999). There is some evidence indicating that certain species of cyanobacteria may act as neurotoxins affecting animals and possibly humans (Falconer, 1999; Rosen, et al., 2001). All of these factors routinely lead to restrictions on freshwater recreation. Additionally, blue-green algae can form mats and block light from penetrating into the water column, limiting photosynthetic activity by macrophytes and reducing biodiversity in the local ecosystem (Gold and Sims, 2005). When cyanobacteria die and decay, decomposition processes reduce and often deplete dissolved oxygen (DO) from the surrounding water, resulting in decreased macrophyte biodiversity and fish kills (Fish et al., 2002; Gold & Sims, 2005; Smith, et al., 2006; Smith et al., 1999; Smith & Schindler, 2009).

#### ***1.1.1 P as Non-Renewable Resource***

Unlike N, which can be extracted from atmospheric sources on an almost unlimited basis, P for use in fertilizer is dependent on the mining of phosphate rock (Cordell, et al., 2009; Neset & Cordell, 2012). Agriculturally, P has no substitute in food production and developments over time such as the Industrial, Sanitation, and Green Revolutions have

conspired to break the global P cycle (Ashley, et al., 2011; Schmid et al.,). Agricultural practices once returned P to soil for eventual re-use through composting, spreading of manure and human “night soils”, and burning of plant matter. With the Industrial Revolution came urbanization and the need for waste management which changed its focus to safe disposal rather than re-use, effectively removing P from natural cycles (Ashley et al., 2011; Neset and Cordell, 2012). Current food production depends on practices developed during the Green Revolution which rely on external fertilizer inputs (Moseley, 2015). These practices are depleting economically-viable phosphate rock reserves with peak P conditions estimated to be reached anywhere between 2030 and sometime in the early 22<sup>nd</sup> century (50-100 years), depending on factors such as fertilizer usage, dietary changes, agricultural practice changes, population increase or decrease, and potential P recycling (Ashley et al., 2011; Cordell et al., 2009; Dawson & Hilton, 2011; Neset & Cordell, 2012; Rittmann, et al., 2011; Steen, 1998; Vaccari & Strigul, 2011). As world P reserves decline, the potential impacts are likely quite serious: rising food prices, food insecurity, and widening inequalities in food access between developing and developed countries (Cordell et al., 2009; Neset & Cordell, 2012; Rosemarin, et al., 2009).

### ***1.1.2 P Recovery and Recycling***

Though debate exists over the exact timeline for depletion of extractable P reserves, agreement on the need for P recapture and re-use is present (Ashley et al., 2011; Dawson and Hilton, 2011; de-Bashan and Bashan, 2004; Neset and Cordell, 2012; Rittmann et al., 2011). A variety of methods exist to capture P in wastewater streams but many extraction methods form sludges that are too toxic to use as fertilizers or create chemical compounds where P is too tightly bound to the precipitation chemical or filtration media to be separated

easily and re-used, resulting in landfilling of the material with its bound P content (de-Bashan and Bashan, 2004; Rittmann et al., 2011). While these methods may prevent P pollution, they do not promote the recycling of P in the environment and thus do little to potentially alleviate a global P shortage.

### **1.1.3** *Nutrient Pollution in the United States*

In the United States, nutrient pollution is a problem that affects millions of people, whether through contamination of drinking water supply or the restriction of water-based recreational activities. Of the 52,000 community drinking water systems across the U.S. serving 290 million people, 78% use ground water and 22% use surface water (U.S. EPA, 2009a). Though data are sparse, the United State Geological Survey found that nearly 73% of domestic wells tested in all 48 states were found to be contaminated with nutrients (DeSimone, et al., 2009). Surface water impairment studies conducted by the U.S. EPA have found that 44% of rivers and streams and 64% of lakes, ponds, and reservoirs suffer from some sort of impairment, though only 16% and 39% of the nation's total surface water acres have been assessed (U.S. EPA, 2009b). Of the top ten causes of impairment, nutrient pollution ranks fifth for rivers and streams, affecting approximately 38,000 miles, whereas for lakes, ponds and reservoirs it ranks third, affecting 1,952,000 acres (U.S. EPA, 2009b). The presence of nutrients in these waters and the potential for eutrophic conditions to create algal blooms often leads to the presence of organic carbon. The combination of organic carbon and disinfection agents used in some water treatment facilities can lead to elevated levels of disinfection by-products in drinking water, which have been linked to health problems in humans (State-EPA Nutrient Innovations Task Group, 2009). The cost of these impacts to the drinking water system is not known precisely (State-EPA Nutrient



Innovations Task Group, 2009). Recreational activities and waterfront aesthetic values are also affected by nutrient pollution (Dodds et al., 2009; Fish et al., 2002; Smith, 2003; Smith and Schindler, 2009; USGS, 1999). Dodds et al (2009) divided economics associated with nutrient pollution into categories, included lost recreational water usage, property value loss for waterfront real estate, spending for the recovery of threatened or endangered species, and the cost of removing contaminants from drinking water. Of these, losses attributed to lakefront property value were the most significant at between \$0.3 – 2.8 billion annually and recreational activity losses at \$0.37 – 1.16 billion annually (Dodds et al., 2009).

## ***1.2 Nutrient Pollution Impact in Vermont***

### ***1.2.1 The Lake Champlain Basin Geography and Demographics***

Vermont's landscape is dominated by three large geographic features. The Green Mountain chain runs north-south down the center of the state, with the Lake Champlain Basin to the West and the Connecticut River valley to the east. The Lake Champlain Basin is an 8,234 square mile watershed spanning two states, Vermont and New York, and the Canadian province of Quebec. The Basin was home to 541,000 U.S. residents at the time of the 2000 census, of which 72% reside in Vermont with 28% in New York; an estimated 30,000 residents in the Quebec portion of the Basin (LCBP, 2003). The U.S population in the Basin has increased approximately 60% since 1950 (LCBP, 2003). The Basin also experiences a seasonal increase in population with the influx of second homeowners, which increases the overall Basin population by 20%. 9,118 of these homes are located on Lake Champlain shoreland areas (LCBP, 2003).

### **1.2.2** *The Lake Champlain Basin Economy*

The economic health of the Lake Champlain Basin is heavily dependent on the natural resources the Basin provides, for agriculture, forestry, tourism, as well as ecosystem services such as nutrient cycling, fresh water, and waste assimilation. (LCBP, 2010). Research indicates that employment in the service sector accounts for 35% of all jobs, followed by trade with 22% and manufacturing with 15%. The overall trend in the 20 years prior to 2010 has seen an increase in service sector and trade jobs, with a decrease in manufacturing (LCBP, 2003).

Tourism is an important aspect of the economy of the Basin. In 2007 it was estimated that visitors spent \$1.6 billion which directly supported 37,000 jobs, or 12% of all jobs, in Vermont (LCBP, 2010). Shoreland towns represent nearly 40% of tourist spending, or \$1.5 billion (LCBP, 2003). A 2006 study found that 45% of visitors to the State cited outdoor recreation as a primary motivator, with 21% responding that aquatic activities were the specific reason for visiting (LCBP, 2010).

Additionally, it was found that the retail and dining sectors were two to three times more dependent on visitors than the national average (LCBP, 2010). However, in a recent book entitled *Don't Go There: The Travel Detective's Essential Guide to the Must-Miss Places of the World*, author Peter Greenberg of NBC's Today Show lists Lake Champlain as a must-avoid place citing frequent water pollution events as the reason (Greenberg, 2009). Potentially losing tourists due to influential reviews such as this would have a potentially significant impact on Vermont's economy.

Seasonal homeowners also account for \$16 million in expenditures on non-durable goods but the continuance of these expenditures depend on a healthy lake (LCBP, 2003). A study in Maine found that a one meter improvement in water clarity improved lakefront property values from \$11 to \$200 per foot of frontage (Michael, et al., 1996). This finding has important ramifications for Vermont's economy.

### **1.2.3** *Lake Champlain's Ecosystem Services*

While tourism contributes directly to jobs in the Basin generally and Vermont in particular, Lake Champlain is also a valuable source of drinking water for over 200,000 residents, with another 4,194 households drawing water from the Lake privately. 137,803 Vermont residents, representing 24% of the Vermont's population, are served by 25 municipal and six commercial drinking water supply stations distributing Lake Champlain water. At wholesale prices for 2001 of \$1.11 per gallon, the total amount of drinking water the Lake provides to Vermont users is estimated at around \$36 million (LCBP, 2003). If nutrient pollution continues at the current rate, or worsens, the cost for potable water could increase.

### **1.2.4** *History of P Pollution in Lake Champlain*

Lake Champlain is a naturally oligotrophic (nutrient-poor) lake, though certain sections do display mesotrophic (intermediate nutrient levels) and in rare cases eutrophic (nutrient rich) tendencies (VT DEC and NY DEC, 2002). Though naturally occurring background levels of P are present in the tributary rivers that feed the Lake, these levels have been influenced over time by anthropogenic influences, most significantly those of agriculture, land conversion, and wastewater treatment plant discharges.

Table 1-1: History of P regulation in Vermont, 1978 – 2016.

Year	Law / Event	Summary	Source
1978	VT Legislature bans P in detergents and cleansers	Ban reduces P loading to the Lake by 50%	(Medalie et al., 2012)
1990	Tribuatriy outflow testing for nutrients	Began to establish baseline nutrient loading data, notably P, for future modeling or loading and reductions	(Medalie et al., 2012)
1990	VT/NY/Quebec begin upgrades of wastewater treatment facilities	Produced significant reductions in point source P contributions.	(Medalie and Smeltzer, 2004)
1993	VT Water Quality Standards adopted	Standards established Lake Segment boundaries, set forth in-Lake criteria for water quality using mass-balance model, established first loading targets for attaining criteria	(Medalie et al., 2012)
2002	VT Dept. of Environmental Conservation submits first TMDL to U.S. EPA	Establishes background levels of P for Lake segments, sets load and wasteload allocations for point / nonpoint P sources	(VT DEC and NY DEC, 2002)
2008	Conservation Law Foundations challenges 2002 TMDL	Challenged TMDL based on wasteload allocations, nonpoint source reduction assurances, margin of safety for targets, inadequate stormwater regulations, failure to consider climate change effects	(Spalding and U.S. EPA, 2011)
2010	Courts grant EPA's voluntary remand	EPA agreed to revise the TMDL addressing CLF's lawsuit challenges	(Spalding and U.S. EPA, 2011)
2016	EPA Issues new TMDL	EPA, working with VT DEC, issues new TMDL for 12 Vermont Lake segments to address P loading	(U.S. EPA, 2016)

Data from the period 1990 to 2000 showed that for all Lake segments, there were at least some years in which in-lake criteria values were exceeded (Medalie and Smeltzer, 2004).

During that time loading from point sources was reduced from 126 mt/yr to 44 mt/yr or 65% largely due to upgrades in P removal techniques used at wastewater treatment facilities (WWTFs). New York State also demonstrated a 33% reduction in point source P loading. However nonpoint source loads remained well above criteria levels during the 1990 to 2000 time period, accounting for 93% of P load at the end of the decade (Medalie and Smeltzer, 2004).

In two recent reports analyzing trends in P loading data from 1990 to 2008 and 1990 to 2009 respectively, the authors came away with similar conclusions (Medalie et al., 2012; Smeltzer, 2009). Though different statistical methods were used to estimate P loading from tributaries to the Lake, the first report concludes that no significant decreasing P trends were found in any segment of the Lake (Smeltzer, 2009) while the second concludes that there has only been a reduction in the magnitude of the upward P loading trend in the Lake (Medalie et al., 2012). A 2005 modeling study conducted by researchers at the University of Vermont created a simple dynamic mass balance model based on P Ecological Indicators such as human population amount and rate of change, land resource use type and change along with associate P runoff coefficients, and in-lake internal loading factors of P such as P-laden sediment re-suspension (Watzin et al., 2005). Most recently it was found that P levels within the Lake have been stable or increasing since 2007 (Lake Champlain Basin Program, 2012).

For Lake segments with relatively high rates of exchange with other Lake segments such as Shelburne Bay and the Main Lake, modeled reductions of 70% in nonpoint source pollution reduced in-lake P levels by +6.4% and -6.3% respectively, while 0%, 25%, and

75% reductions in Mississquoi Bay resulted in +15%, 0%, and -33% reductions respectively (Watzin et al., 2005). It is clear from this model that significant changes in in-lake P levels can only be brought about by drastic reductions in P loading to the Lake. To date, considerable investment has been made to attempt to improve water quality in Lake Champlain. According to the Lake Champlain Basin Program Report in 2010, \$40 million has been spent on upgrading P removal practices at 35 wastewater treatment facilities in Vermont (LCBP, 2010). Additionally Vermont's Clean and Clear Action Plan, subsequently known as the Ecosystem Restoration Program and now titled the Clean Water Initiative, injected nearly \$100 million since 2004 in state and federal funds to support a wide range of programs designed primarily to reduce nonpoint sources of pollution (LCBP, 2010).

Recently the National Science Foundation awarded the University of Vermont nearly \$20 million for research into climate change adaptation in the Lake Champlain Basin (Vermont EPSCoR, 2012). On a policy level, the State of Vermont signed Act 64, known as Vermont's Clean Water act, signed into a law a number of changes designed to reduce point and nonpoint sources of pollution, as well as provide for their funding through the establishment of a Clean Water Fund (State of Vermont, 2015). Despite these numerous programs and funding dollars, the U.S. Environmental Protection Agency formally disapproved the State of Vermont's action plan for cleaning up the Lake and took control of the program in 2011 (Spalding and U.S. EPA, 2011).

Subsequent to the 2011 disapproval of the TMDL, the State and EPA worked to develop a new TMDL, issuing that document in 2016 (U.S. EPA, 2016).

### ***1.3 P Discharge and Removal Regulations***

#### ***1.3.1 National Pollutant Discharge Elimination System (NPDES) Total Maximum Daily Load (TMDL) Program***

The United States government first started to address the problem of polluted waterways with the Federal Water Pollution Control Act of 1948, which was significantly amended in 1972 to become known as the Clean Water Act (CWA) (Tchobanoglous and Burton, 1981; U.S. EPA, 2000a; US EPA, 2012). The Act established NPDES, which set permitted levels for each discharger of a potential pollutant. Amendments to the CWA in 1987, 1993, and 1994 set regulations for solids control, metals contamination, pathogen content, and combined sewer overflow controls (Tchobanoglous and Burton, 1981). In 2000 Section 303(d) was added to the CWA requiring states to develop prioritized lists of polluted or threatened water bodies and establish the total maximum daily load (TMDL) that a water body can receive and still meet basic water quality standards (Copeland, 2010; U.S. EPA, 2012). By 2012 over 49,000 TMDLs had been defined throughout the United States for over 45,000 impaired water bodies (U.S. EPA, 2012). Nutrients are currently the third most common cause of 303(d) impairment, affecting nearly 7,000 water bodies (U.S. EPA, 2012).

#### ***1.3.2 Lake Champlain P Total Maximum Daily Load (TMDL):***

In 2002, Vermont's Department of Environmental Conservation submitted its first TMDL to the U.S. EPA. The TMDL document outlines a plan of action for achieving P reductions for Lake Champlain. Modeling was conducted by the U.S. Army Corps of Engineers using the BATHTUB model to determine background P loading levels for each Lake segment, as well as establish how circulation patterns within the Lake would affect hydrologic mixing and P diffusion throughout the water column (VT DEC and NY DEC, 2002). Using

background levels and WWTF emission data, an annual target load of 439 mt/yr was set for the Lake, 319 mt/yr of which was attributed to VT and QC, with the remaining 120 mt/yr from NY. These amounts represent a 30% reduction from the 1991 reference year levels. For Vermont, point sources must be reduced from the 1991 level of 121.1 mt/yr to 41.1 mt/yr while nonpoint sources must be reduced from 293.1 mt/yr to 277.5 mt/yr (VT DEC and NY DEC, 2002).

The Regional U.S. EPA office approved of the Lake Champlain P TMDL on November 4, 2002 (Spalding and U.S. EPA, 2011). A revision to the TMDL was issued in 2010 refining the need and levels of funding for various individual aspects of the plan (VT ANR, 2010). The approval was challenged by the Conservation Law Foundation on October 28, 2008 on the grounds that the TMDL did not have sufficiently stringent wasteload allocations in conjunction with a lack of assurance that nonpoint source reductions would occur, there was an inadequate margin of safety in the reduction target allocations that were mandated, that stormwater regulations were inadequate and insufficient or inadequate data had been used to develop them, and that there was a failure to consider the effects of climate change on tributary flows and P loading (Spalding and U.S. EPA, 2011). In August 2010 courts granted the U.S. EPA's motion for voluntary remand to reconsider its findings on the Lake Champlain TMDL, originally to be completed by early 2011. The final, most recent TMDL was completed in 2016 and addresses the concerns of CLF (U.S. EPA, 2016).

The U.S. EPA found that with respect to stormwater wasteload allocations and calculation of loading capacity based on the 1991 hydrologic year and climate change considerations,



the TMDL used the best available scientifically sound data and was found to be adequate according to TMDL guidelines (Spalding and U.S. EPA, 2011).

However, EPA found that the TMDL did not create adequate assurances with respect to the amount of P reduction expected from wasteload allocations – essentially finding that tolerances for concentrations were too high and that ratio of dissolved to particulate phosphorus did not decrease as anticipated. Additionally, EPA found that load allocation reductions attributable to VT DEC programs to address stormwater were inadequate to address the anticipated 80.5 mt/yr reduction required (Spalding and U.S. EPA, 2011). For the revised 2016 TMDL, additional requirements were created to ensure that load allocation reductions would be made. These load reductions rely on reductions and no-net increase in P from nonpoint sources (U.S. EPA, 2016).

#### ***1.4 TMDL Compliance***

##### ***1.4.1 Nutrient Trading***

Based on decades of analysis of costs associated with water quality improvement, U.S. EPA estimates that up to \$900 million could be saved annually in the U.S. using flexible approaches to pollution mitigation, including nutrient trading (U.S. EPA, 2003, 2001, 2000a, 1996). Nutrient trading works by allowing individual polluters to trade among themselves to determine who pollutes less or more (Shabman & Stephenson, 2007; Shortle, 2013; Shortle & Horan, 2006). A total cap on pollution levels is established (the TMDL in this case) and trading reallocates pollution levels among sources. If one source can economically decrease its pollution emissions, it can then sell credits to another emitter if their respective cost of abatement is economically unfeasible for them to address

(Woodward, et al., 2007). This practice is not novel as it is based on established protocols for trading air pollution emissions credits as set forth by the Kyoto Protocol on Climate Change (Woodward et al., 2007). As of 2005 there were as many as 69 different water quality related nutrient trading initiatives in the United States (Morgan and Wolverton, 2005). Though the framework for these programs is in place, many of them are non-functioning or hindered by lack of true regulatory teeth (King, 2005). However without any substantial regulatory requirement for P discharge limits or removal, which would increase the demand for credits, there is no incentive to trade. For instance, in 2003 when Maryland instituted a ‘flush tax’ on municipal water and sewer ratepayers to subsidize state of the art treatment technologies at wastewater treatment plants, the measure effectively eliminated demand for nutrient trading credits by removing the need for WWTPs (with their relatively high discharge treatment costs) to purchase allowances from nonpoint sources (with their relatively low discharge treatment costs) (King, 2005). Though TMDLs may ultimately be the driver of demand for nutrient reduction, an adequate supply of long-term cost-effective nutrient reduction measures must also be in place to ensure that sources of pollution take advantage of them.

#### **1.4.2** *Current Nutrient Pollution Reduction Measures*

Nutrient pollution reduction measures are classified according to source type (point versus nonpoint), which are then attributed to municipal, agricultural, industrial, or residential. Conventional wastewater P removal practices involve complex and costly chemical and/or biological processes that often require large amounts of energy, work-intensive operation and maintenance, and generate sludge, which presents a waste disposal issue (Dijk et al., 2012; Jiang, et al., 2005; Tchobanoglous & Burton, 1981).

#### ***1.4.2.1 Municipal Wastewater Facilities and P Removal***

Chemical removal of P involves the addition of ferric salts, alum, or lime to wastewater, which in turn generates a P-rich sludge that must be disposed of off-site. Biological processes to remove P include the Enhanced Biological P Removal (EBPR), the Anoxic/Oxic (A/O), and the Anaerobic/Anoxic/Oxic processes which rely on successive sedimentation (Cite). Using both chemical and biological methods of P removal, P-rich sludge is accumulated and must be stabilized biologically through composting or anaerobic or aerobic digestion, chemically through lime addition, or thermally through drying or incineration for volume reduction and pathogen inactivation (Jiang et al., 2005; Tchobanoglous and Burton, 1981; U.S. EPA, 2013a). Sludge is typically land-applied as it has value as a fertilizer; however, because of its origin, sludge may contain excessive heavy metals, pathogens, household chemicals, and pharmaceuticals. For this reason land application is often debated (Dijk et al., 2012).

The costs of P removal from wastewater can be considerable, and varies with the P discharge standard for effluent wastewater. One study showed that for two biological processes and one chemical process, meeting discharge limits of 1 mg/L P could potentially require capital investments in the hundreds of millions of dollars. When the standard increases to 0.5 mg/L P, costs for chemical removal of P increase tenfold annually while costs for biological processes nearly triple (Jiang et al., 2005).

#### ***1.4.2.2 On-Site Residential Wastewater Treatment and P Removal***

25-30% of the global population relies on septic systems, usually consisting of a septic tank and a gravity-fed drain field (Dijk et al., 2012). Soils in the drain field are generally considered to be naturally P adsorbing materials, however this depends on soil type and P

saturation over years of use (Drizo, 2002; Postma, et al., 1992; Robertson, et al., 2005). Additionally, such systems are often poorly maintained and suffer failure rates from 10 – 20% (U.S. EPA, 2000b). Some studies report that these failures contribute as much as 20% of P load in rural watersheds (Dijk et al., 2012; Postma et al., 1992; Withers, et al., 2011). This situation is much worse in developing countries where residential building codes are unenforced or non-existent, leading to widespread use of rivers as open sewers (Dijk et al., 2012).

The need to remove P from residential wastewater has resulted in the rise of several commercial solutions which specifically target P. They can be divided into passive and active removal methods.

The first passive method product is a product made from calcium silicate refined and formulated to specific grain sizes to effect P removal via adsorption to calcium. These materials go under the names Polonite and Sorbulite and are predominantly employed in Europe, building on the research of Cucarella & Renman (2006) (Renman, 2015). The second passive method product was developed by a team of University of Waterloo researchers in the late 1990s and relies on the use of basic oxygen furnace (BOF) slag primarily to effect P removal (Smyth & Blowes, 2004; Smyth, et al., 2004). This media has a high calcium content as well and the product goes under the name Phosphex (US Patent #5,876,606, Canada #2,190,033, UK #2,306,954). This system was re-licensed in the United States and goes by the name of PhosRid (Lombardo, 2004). The third product is known as PhosphoReduc and was developed based on research conducted by Aleksandra Drizo (Drizo, et al., 1999; Drizo, et al., 2002, Drizo, et al., 2006; Bird & Drizo, 2010; Weber, et. al., 2007; Lee, et al., 2010) and developed in collaboration with Hugo Picard (Drizo & Picard, 2014, US Patent # 8,721,885). The system relies on the use of steel slags which are tested for their respective

Fe and Ca content, then used in a multi-compartment filter system to effect P removal via adsorption to Ca and Fe oxides.

Additionally there are emerging technologies which make use of electro chemical processes which require the use of iron electrodes submerged in wastewater. A current is passed through them, resulting in iron ion erosion, which then bonds with P forming  $\text{FePO}_4$  (Otowa, 2014).

#### **1.4.2.3 *Agricultural Best Management Practices for P Removal***

Numerous practices exist to manage runoff from agricultural lands. These practices are commonly referred to as ‘Best Management Practices’ in North America though in Vermont they are currently termed ‘Required Agricultural Practices’ (VT AAFM, 2016). In Europe and other parts of the world they are referred to as Good Agricultural Practices (GAPs). They were developed in 1950s as the conservation measure to combat soil erosion and were implemented as soil remediation practices for two decades prior to the first awareness and recognition that non-point source (NPS) phosphorus from soil run-off and erosion, fertilizer and livestock runoff results in eutrophication of water bodies in 1970s (Logan, 1993). Sharpley, et. al., (2006) defines them as practices that include soil and water conservation practices, as well as other management techniques and social actions developed for environmental protection. They can take the form of nutrient management plans, changes in farming operations, or more structural practices like manure storage systems and runoff prevention practices (Sharpley, et al., 2006). Most agricultural BMPs for P pollution reduction are considered either source or transport BMPs. Source BMPs minimize the loss of P at the source whereas transport BMPs minimize the potential for P to be transported off-site (Sharpley, et al., 2006).

#### ***1.4.2.4 Urban Stormwater Runoff Control Measures***

As the U.S. urbanized, combined sewer systems that collect both residential wastewater as well as stormwater runoff were typically installed as they were less expensive to build than separate sewage and stormwater systems (U.S. EPA, 2011a, 2011b). These systems are routed to wastewater treatment facilities where the water is treated and discharged. During heavy rains, the volume of runoff can overwhelm the capacity of a WWTF, causing discharge of untreated waters, known as a combined sewer overflow (CSO). CSOs have become a serious source of pollution in the U.S. (U.S. EPA, 2011a, 2011b, 2011c).

There are also separate storm sewer systems, sometimes referred to as municipal separate storm sewer systems (MS4s), though this is technically a regulatory designation and does not necessarily cover all separate storm sewer systems. These systems do not commingle stormwater runoff with sewage but instead collect runoff into closed catch basins and pipes and discharge that water into local water bodies (often without treatment). Discharge from separate storm sewer systems represents an important vector for water pollution (Kim, et al., 2005; LUO et al., 2009; Pitt et al., 2004; Walker, et al., 1999)

During the past 10-15 years, stormwater best management practices (BMPs) have been designed and implemented to reduce pollutant loads from developed lands. There are a variety of practices as outlined in the U.S. EPA's 'National Menu of Stormwater Management Best Practices.' BMPs provide storage for excess runoff volumes and provide for treatment of organic matter, suspended solids, metals, oils, bacteria, and nutrients (U.S. EPA, 2013b). However, performance for P varies widely among BMP types (Ballesterio, et al., 2012) As a result many companies have developed new products specifically designed

to reduce P in stormwater. However, the proprietary products (e.g., (“Contech Stormwater Management StormFilter” 2013, “Hydro International - Stormwater” 2013, “Imbrium Systems” 2013, “Imbrium Systems News Release” 2009) may not always have undergone significant, independent scientific evaluation to determine their efficacy (Contech, 2013; Glennon, et al., 2006; HydroInternational, 2013; Imbrium Systems, 2013; Imbrium Systems, 2009). Costs for these systems is regional and largely un-quantified (U.S. EPA, n.d.).

### ***1.5 Use of Industrial and Natural By-products for P Removal***

The research on the efficacy of various natural and industrial by-products for removing P was initiated in the 1990s, primarily focusing on alternative substrates in constructed treatment wetlands. Yamada et al. (1986) investigated steel slag aggregates and found that the adsorption of dissolved inorganic phosphate to this material depended on pH, temperature, the concentration of the coexisting salt, and the porosity of the slag matrix. In addition, Yamada et al. (1986) found that adsorption sites on the slag were coincident with calcium, magnesium, aluminum and silicon compounds and that dissolution of phosphate from adsorption sites was small.

Drizo, et al. (1999) first investigated the use of locally available industrial by-products in the UK, testing bauxite, shale, burnt oil shale, limestone, zeolite, light expanded clay aggregates (LECA) and fly ash. Further refinement of substrate material selection was provided by Zhu, et al. (1997) in research showing that the P uptake capacity of five lightweight aggregates was found to depend strongly on the Mg, Ca, Fe, and Al metal ion content of the aggregates, with Ca ions having the strongest correlation with P sorption

capacity. Johansson (1999) investigated crystalline and amorphous blast furnace slag for its suitability for P sorption; laboratory column studies showed that up to 95% of P was removed under conditions of intermittent loading (0.6 L P solution every second hour) using both coarse and fine slag mixtures.

Johansson (2006) reviewed 35 different studies that examined the P-sorption capacity of 16 different materials. Of these 16, ten were the natural materials, including limestone, opoka, wollastonite, bauxite, zeolite, laterite, marl, spodosols, maerl, and shells or shell sands; four were industrial by-products such as blast furnace slag, electric arc furnace slag, burnt oil shale, and fly ash; and two were the man-made materials Filtralite and light-weight aggregates (LWA) also known as LECA. Data normalization for P-sorption capacity between these studies was problematic due to the variety of experiment types (batch, column, or field studies), the variation between material types, and the variation in P-sorption capacity measurement methods (calculated vs. measured). Preliminary conclusions drawn from this study showed that wollastonite, steel slags, and the man-made materials showed the most promising P-sorption capacities (Johansson-Westholm, 2006). Notably, the author specifically mentions the need for field-based investigation and the use of real in-situ wastewaters as most of the studies used synthetic wastewaters in lab environments (Johansson-Westholm, 2006).

Based in part on the promising P adsorption results shown by industrial by-products (IBPs), Chazarenc, et al. (2008) conducted a review of 46 papers specifically investigating IBPs. Most (49%) were simple batch studies, 38% were column studies, and only 13% were systems tested in the field. As with the Johansson-Westholm (2006) study, normalization



of data was difficult given the various conditions under which experiments were conducted but several field-scale design considerations were drawn based on observations from the various lab and field studies. It was found that there was generally a tendency to misestimate the amount of material to use (Chazarenc et al., 2008; Hylander, et al., 2006). Additionally, the use of IBPs as the primary substrate in constructed wetlands (CWs) was problematic due to some difficulties with plant establishment in the high pH environment created by IBPs during P-sorption, as well as the potential difficulty in excavating and replacing material upon P-saturation with respect to destruction of established plant communities (Chazarenc et al., 2008; Vohla, et al., 2011; Vohla, Pöldvere, et al., 2005). Typically macrophytes in CWs are thought to provide some nutrient uptake, as well as increase aeration in sub-surface areas and provide root zones for microbial communities (Langergraber & Haberl, 2001; Vymazal, 2011; Vymazal, et al., 2006). However Tanner (2001) found that plant uptake of nutrients, in particular P, was only initially significant when compared with unplanted systems – over the long-term, P-uptake by plants was found to be an insignificant factor in CWs. For treatments systems, in particular stormwater BMPs that are to be planted, using IBPs as the primary substrate could create issues.

Chazarenc, et. al., make the following design recommendations based on their extensive review of IBPs used in CWs

- Field-testing the specific material to be used (not laboratory batch-testing) will assist in more accurately gauging the amount of material needed to treat design flows (Drizo et al., 2002).

- IBP filters should have some form of pre-treatment to avoid observed problems of bio-film development on substrate media, which clogs media pores and leads to reduce P uptake (Cameron, et al., 2003; Chan, et al., 2008; Shilton et al., 2006; Vohla et al., 2005).
- Saturated flow was preferable, primarily because it promoted a contact time between wastewater and substrate media of around 24 hours (Chazarenc et al., 2008).

The only long-term study of IBPs in wastewater treatment found that the general P-saturation level of 1.2 kg P / ton of material is a design consideration with respect to replacement or rejuvenation, but this number is only applicable to the ‘melter’ slag used in this study (Shilton et al., 2006).

A review of previous studies by Vohla et al. (2011) further confirms the findings by Chazarenc et al. – that though certain materials appear promising in the lab, few have been studied in the field and this represents the most pressing area of need for the use of any material to be used in P reduction from wastewater streams. Vohla (2011) does highlight that certain steel slags show the highest P-removal capacities, though data were not normalized.

#### ***1.5.1 Field Studies on P-sorption capacity of Industrial Byproducts***

Of all the research conducted on P- sorption capacity, very few have been field-based. A survey of work done between 1991 and 2002 found that of 35 different studies, only 11 had some field-based research (Johansson-Westholm, 2006). A review focusing solely on IBPs found that between 1986 and 2008, of 46 studies reviewed only seven or 13% had

some field-based research (Chazarenc et al., 2008). In 2011 a review published of natural, IBP, and man-made materials showed that 19 of 79 (fewer than 1/3<sup>rd</sup>) studies used data from field-based research (Vohla et al., 2011). Based on the findings from this study, Vohla et al., (2011) state in the conclusion that ‘well monitored long-term experiments in full-size systems are needed’.

### **1.5.2** *Field Studies on Steel Slag Aggregates as Potential Filtration Material for P Removal*

Steel slag is a byproduct of steel manufacturing when hot iron (basic oxygen furnace (BOF)) or steel scrap (electric arc furnace (EAF)) are melted and combined with lime to act as a fluxing agent (National Slag Association, 2016). The lime combines with silicates, aluminum oxides, magnesium oxides, manganese oxides, and ferrites to form steel slag (National Slag Association, 2016). In a basic oxygen furnace oxygen is blown into the furnace to oxidize carbon and other unwanted elements in the molten iron, with other fluxes added to yield higher-quality steel (National Slag Association, 2016). In an electric arc furnace, material is heated to a liquid state using electrodes (National Slag Association, 2016). An additional type of steel production using a blast furnace (BF) to heat iron ore or pellets along with coke and a fluxing agent (either limestone or dolomite) produces blast furnace slag. Finally, there is one other type of slag, known as iron-melter slag (IM), from New Zealand which is formed when the iron-rich sand goes through a direct reduction process (Bourke, et al., 2005). This type of steel production is endemic to New Zealand as the iron-rich sand is not found anywhere else.

Each of these slags have slightly different chemical characteristics, which can affect their potential to remove P from wastewater (Barca, 2014). These chemical compositions are shown in Table 1-2.

Table 1-2: Chemical composition of the major types of slag produced in steel making. Compositions for BF, BOF, and EAF slags are from Proctor (2000).

<b>Element</b>	<b>BF (mg/kg)</b>	<b>BOF (mg/kg)</b>	<b>EAF (mg/kg)</b>
Al	41245	23841	35009
Ca	273855	280135	250653
Cr	132	1271	3046
Fe	17355	184300	190211
Mg	69991	55318	54460
Mn	5527	32853	39400
Si	170064	59683	74524
Ti	ND	7.2	11

To date, field studies on steel slag based filtration have been sparse (Drizo, et al., 2002, Drizo, et al., 2006; Chazarenc et al., 2008; Cucarella & Renman, 2006; Johansson-Westholm, 2006; Renman, 2011; Vohla et al., 2011). Most of these studies were published beginning in the early 2000s, though some describe experiments dating back into the late 1990s. Baker et al. (1997) evaluated P treatment performance of blast furnace (BF) steel slag in two field trials, finding that for treatment of primarily treated municipal wastewater, BF slag removed as much 95% of both DRP and TP over a period of four months (Baker, et al., 1997).

In 2002 Smyth, et al., published the results of two field trials conducted in 1999 (Smyth and Blowes, 2002). BF slag treating residential septic tank effluent was found to reduce P to less than 0.05 mg/L, though influent concentrations were not measured, nor were loading rates or hydraulic residence times (HRTs) given. It was also found that BF slag treating communal municipal wastewater with an influent concentration of 3.75 mg/L P with a HRT of 24 hours reduced P to less than 0.05 mg/L, though it was noted that for both trials elevated pH, at times greater than 12, was found in the effluent (Smyth and Blowes, 2002). A field trial conducted in Canada treating municipal wastewater from a primary treatment lagoon found that BF slag filters removed 99.40% of P when influent P concentration was 0.91 mg/L (Cameron et al., 2003). However, Cameron et. al. (2003) found that removal efficiency decreased when the filters were fed with effluent from a free-surface constructed wetland and a vegetated buffer strip to 54.05% and 59.21% respectively, indicating that overall removal efficiency decreases with lower P concentrations (i.e. cleaner water).

A New Zealand study found that over the course of a five year period from 1993 to 1998 a slag filter treating effluent from a municipal wastewater lagoon reduced P concentration by nearly 77%; removal efficiency only decreased when adsorption sites on the slag surface began to saturate at a level of approximately 1.23 kg P / ton of material (Shilton et al., 2006). A 2006 study found that when compared to gravel and “clinker” (a by-product of cement production), BF slag treating domestic wastewater consistently had the lowest effluent P concentration of all three materials (Calder, et al., 2006). A Turkish study which also investigated BF slag found that average removal efficiencies for DRP and TP were

44% and 46% respectively with an hydraulic residence time (HRT) of approximately 4-6 days over a 12 month monitoring period (Asuman et al., 2007).

Recently, numerous field, pilot, and demonstration projects have been conducted by Dr. Aleksandra Drizo and collaborators at the University of Vermont investigating the use of electric arc furnace (EAF) steel slag treating agricultural point and nonpoint source P and solids pollution reduction (Bird & Drizo, 2010; Lee et al., 2010; Weber, et al., 2007). Weber et al. (2007) conducted a study at the University of Vermont Constructed Wetland Research Center which found that EAF steel slag filters, used in combination with a constructed wetland treating a dairy operation feedlot runoff and milking parlor wash water, reduced P concentration in effluent by 75% on average, with P concentrations in influent wastewater ranging from 10.7 to 55.7 mg / L P. When used as stand-alone filters treating untreated wastewater from the same source, the filters were found to remove 72% of P with 12-24 hour retention times.

Evidence was also found that EAF steel slag's P retention capacity could be rejuvenated by up to 49.5% if slag was given a "resting period" prior to reaching saturation (Drizo et al., 2008). Lee, et al., (2010) confirmed these findings in a study on the effects of adding an EAF steel slag filter to the effluent stream of various constructed wetlands, finding that P removal efficiencies approached 99% for all systems with slag filters. Systems constructed without slag filters averaged between 50-75% DRP removal for all samples and filter types.

Bird & Drizo (2010) studied the effects of in-series design on filter performance, finding that P removal efficiency for a single filter was around 42% versus 69% for two filters in

a series. The researchers were also able to study the effect of total suspended solids (TSS) mass loading and filter clogging performance, finding that influent TSS levels averaging 3395 mg/L, which exceed the EPA recommended guideline for TSS loading when attempting to maximize P removal by nearly 34 fold, clogged the filters after 58 days. Despite this non-ideal operation parameter, the filters still achieved 38% removal efficiency.

Although EAF steel slag contains certain metals which are of concern if leached, some studies have reported that there are no significant environmental problems associated with metals leaching in slags (Steel Slag Coalition, 2007; Australasian Slag Association, Inc., 2006). Other reviews have indicated that ferrous-slags (like EAF steel slag) could be a source of environmental contamination, depending on its environmental context (Piatak, et al., 2015). However, preliminary data based on several pilot studies in Vermont and across the U.S. Midwest suggest that EAF steel slag sourced from Quebec does not leach metals (Drizo, pers. comm., December, 2016).

Research has shown that modified steel slag aggregate filtration material has a potential for use as a soil amendment in forestry, agriculture, and acid mine drainage as a slow release P fertilizer with no leaching of metals after it has reached P saturation in a filter application, thus providing an additional use for EAF steel slag that has reached its saturation potential for P removal (Bird & Drizo, 2010; Hylander et al., 2006; Hylander & Simán, 2001).

This thesis research deals primarily with the application of EAF steel slag in two filtration settings – at a wastewater treatment facility treating cheese factory wash water lagoon

effluent and as a biofilter soil media amendment to enhance P removal in a stormwater BMP.

### ***1.6 The Need for Innovative P-Removal Technologies***

The need for innovative P removal technologies to mitigate P pollution is growing both nationally and internationally. Long-term water quality monitoring and evaluation of field-based treatment systems is a prerequisite to the development of a full-scale application. Dr. Drizo and collaborators at the University of Vermont Constructed Wetland Research Center demonstrated the efficiency of the EAF steel slag filter technology in P, solids, and *E. coli* reduction in a variety of field-based pilot projects between 2004 and 2011 (Bird & Drizo, 2010; Drizo et al., 2008; Lee et al., 2010; Weber et al., 2007).

In Vermont, programs like UVM's Food Systems Initiative seek to strengthen the viability of regional food systems production while creating jobs – dairy and dairy products sold from Vermont farms accounted for over \$500 million in sales in 2012 (USDA-NASS, 2013).

While this figures is important for the State's economy, the production of dairy products, especially cheese, generate wastewaters containing high P concentrations.

### ***1.7 Study Site***

In 2010, Swan Valley Cheese of Vermont (S.V.C.), a subsidiary of national dairy-product distributor Lotito Foods, received over \$380,000 in financial assistance from the Vermont Economic Progress Council to start production of ricotta and mozzarella cheese in the former Via Cheese factory building in Swanton, VT. In 2012, S.V.C. employed 22 people and was generating approximately 40,000 gallons of cheese production related wash water,



laden with whey residue, organic and inorganic solids, and cleaning products. S.V.C. maintains a primary treatment aeration lagoon for initial wastewater treatment which reduces P, N, TSS, and BOD levels, but discharge wastewaters with an approximate annual total of 3,000 lbs. of total P, costing S.V.C. around \$6,000.00 per month for P removal via alum precipitation and sludge landfilling provided by the Swanton Municipal Wastewater Treatment Facility (SMWTF). Both entities are located in the Mississquoi Bay watershed of Lake Champlain, known to have some of the highest concentrations of P in the Lake (Lake Champlain Basin Program, 2012).

## 1.8 References

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## 2 PHOSPHORUS AND SUSPENDED SOLIDS REMOVAL FROM CHEESE-FACTORY WASHWATER USING AN ELECTRIC ARC FURNACE (EAF) STEEL SLAG FILTER

### 2.1 ABSTRACT

An electric arc furnace (EAF) steel slag filter was investigated for its effect on Total Phosphorus (TP), Dissolved Reactive Phosphorus (DRP), and Total Suspended Solids (TSS), from wastewater associated with cheese production at a facility in Swanton, VT. The primary objective was to conduct a field-test of the filter's effect on P in wastewater. Secondary objectives were to 1) test the filter's effect on TSS, 2) investigate the effect of using multiple filter compartments filled with different media on P and TSS and 3) to investigate a method for reducing pH of wastewater passed through the filter.

One concrete 3,000 gallon (11.35 m<sup>3</sup>) filter divided into three compartments was put into operation in December, 2011. Influent was first fed into Compartment 1 (271 gallon – 1.02 m<sup>3</sup>), which was empty and intended for solids settling. Compartment 2 (2046 gallon – 7.75 m<sup>3</sup>) was filled with granulated EAF steel slag. The final compartment, Compartment 3 (439 gallon – 1.66 m<sup>3</sup>), was filled with Eastern White Cedar (*Thuja occidentalis*) wood chips contained in wire-mesh gabion baskets and planted during summer months with *Juncus effusus* and *Typha latifolia*. Compartment 3 was intended to reduce pH levels in wastewater after contact with EAF steel slag.

The filter was fed wastewater from the cheese production facility's pre-treatment aerated lagoon. Wastewater consisted of process and cheese production machine wash water – a mixture of whey waste and cleaning products. Influent rates varied greatly during filter operation due to cheese production schedules at the plant. As a result, hydraulic residence times were variable and often in excess of 24 hours during the research period.

The median influent TP, DRP, and TSS concentrations during the filter's operation were 24.00 mg/L  $\pm$  6.69 mg/L (n = 23), 19.52 mg/L  $\pm$  5.06 mg/L (n = 21), and 53.81 mg/L  $\pm$  44.54 mg/L (n = 22) respectively during the 152 day operation period. Median filter TP, DRP, and TSS effluent (all compartments) during this period was 1.00 mg/L  $\pm$  2.54 mg/L (n=23), 0.65 mg/L  $\pm$  5.04 mg/L (n=21), and 25.69 mg/L  $\pm$  49.28 mg/L (n=22) respectively. Removal efficiencies for TP, DRP, and TSS concentrations from influent to effluent during this period were 95.83%, 96.65%, and 52.25%. Statistical analysis shows that TP, DRP, and TSS influent concentrations were significantly different from effluent concentrations. Differences in removal efficiency between compartments was variable, however removal rates were nearly always highest from Compartment 2 for all analytes.

The filter's effect on pH was analyzed and a statistically significant difference was shown between Compartment 2 and Compartment 3, indicating that the wood chips reduced median Compartment 2 pH from 11.17  $\pm$  0.40 to 10.12  $\pm$  1.55. However, filter effluent from Compartment 3 at 10.12  $\pm$  1.55 was statistically significantly higher than influent

pH at  $7.87 \pm 0.25$ .

This research indicates that EAF steel slag filters can be effective at removing TP and DRP from cheese production facility wash water under variable hydraulic loading conditions. This research also indicates that the use of wood chips to reduce the pH of wastewater post-EAF steel slag contact may be effective in meeting pH discharge standards.

The EAF steel slag filter investigated during this research was designed under the guidance of Dr. Aleksandra Drizo in association with her company PhosphoReduc, LLC (<http://phosphoreduc.org/>). Dr. Drizo's research and design of innovative P-removal filters with PhosphoReduc draws on over decade of experience in the field with solutions targeted at municipal, industrial, agricultural, and residential wastewater treatment.

## **2.2 KEYWORDS:**

Electric arc furnace steel slag, phosphorus, whey wastewater, cheese production, pH, total suspended solids filtration

## **2.3 INTRODUCTION**

### **2.3.1 *Eutrophication and Wastewater Concerns in the Lake Champlain Basin***

Nutrient loading from human activities, or eutrophication, has been recognized as a leading cause of water quality impairment for over 50 years (Conley et al., 2009; Ryther and Dunstan, 1971; Schelske, 2009; Schindler, 1974). In freshwater ecosystems, excess P in the system can result in blooms of primary photosynthetic producers in the cyanobacteria family, commonly known as blue-green algae (Gold & Sims, 2005; Smith, 1982; Smith, et al., 1999; USGS, 1999). Surface water impairment studies conducted by the U.S. EPA have found that 44% of rivers and streams and 64% of lakes, ponds, and reservoirs suffer from some sort of impairment, though only 16% and 39% of the nation's total surface water acres have been assessed (U.S. EPA, 2009b).

Lake Champlain is a large lake in the northeast region of the United States of America between the states of Vermont and New York and bordering the southern portion of the

province of Quebec. Over the past several decades it has become severely impacted by P pollution from point and non-point sources ((LCBP, 2010; Watzin et al., 2005). Impairment of the Lake has a direct economic impact on the economy of the region, in particular in Vermont, the most populous area within the Lake Champlain Basin. Tourism in Vermont alone accounts for nearly \$1.5 billion in revenue, with residents of the state living within 35 miles of the Lake spending nearly \$118 million on water-based recreational activities in 1998 (LCBP, 2003; VT ANR, 1997). Additionally, second homeowners living along the Lake's shoreline account for \$16 million in annual expenditures on non-durable goods; loss of these revenue streams due to Lake Champlain's eutrophication could potentially have a serious impact on the Vermont economy (LCBP, 2003).

P pollution in Lake Champlain is attributable to both point and non-point sources. Point sources comprise discharges from wastewater treatment facilities (WWTFs). In 1990, Vermont, along with New York and the province of Quebec began to upgrade their respective WWTFs, resulting in significant reductions in related point-source contributions of P (Medalie and Smeltzer, 2004). In 1993, the Vermont Water Quality Standards were adopted and endorsed by Vermont, as well as the state of New York and the province of Quebec. These standards established Lake segment boundaries, set forth in-lake criteria for water quality using a mass-balance model, and established the first loading targets consistent with attaining the in-lake criteria (Medalie and Smeltzer, 2004). In 1993, criteria for in-Lake P concentrations were set between 0.010 to 0.025 mg/L P depending on Lake segment, with a total mass load reduction of 192 metric tons per year

(mt/yr), later revised to 202 mt/yr (Medalie, et al., 2012). From 1990 to 2000, loading from point sources was reduced from 126 mt/yr to 44 mt/yr or 65% largely due to upgrades in P removal techniques at WWTFs. \$40 million has been spent on upgrading P removal practices at 35 wastewater treatment facilities in Vermont (LCBP, 2010).

The 2002 TMDL relied on limits for WWTF P concentration for effluents at 0.8 mg/L P, though facilities permitted before 1991 and discharging less than 0.2 million gallons per day (mgd) were exempt from this rule – a default value of 5.0 mg/L was used for those facilities for planning purposes (VT DEC and NY DEC, 2002). This wasteload allocation, among other aspects of the TMDL, was challenged by the Conservation Law Foundation, a challenge which was upheld in August 2010 (Spalding and U.S. EPA, 2011). The U.S. Environmental Protection Agency formally disapproved the State of Vermont's action plan, or Total Maximum Daily Load (TMDL) allocation, for cleaning up the Lake and took control of the program in 2011 (Spalding and U.S. EPA, 2011).

This has led to the creation of a new TMDL by the U.S. EPA. The draft TMDL, issued in August, 2015, does define additional reductions to be made at WWTFs throughout the Basin, which depend on the relative ratio of point source to non-point source loading for each particular Lake segment and the capacity of the WWTFs (in gallons/day) (US EPA, 2015). Costs for WWTPs associated with achieving low effluent P concentrations (0.1 – 0.2 mg/L) can be substantial (Jiang, et al., 2005; Tetra Tech, 2013; U.S. EPA, 2007). For this reason, exploring cost-effective methods to reduce P in WWTF effluent will aid in achieving compliance with the new TMDL.



### **2.3.2** *The Vermont Dairy Industry*

In Vermont between 70-80% of all agricultural sales are related to the dairy industry (Parsons, 2010). This industry is estimated to be worth \$2.2B in Vermont, with dairy sales accounting for approximately \$1.3B annually (Galloway, 2015). This represents a significant amount of Vermont's economic activity, yet due to rising costs associated, in part, with conforming to environmental regulations concerning phosphorus in runoff from farms and production of value-added products such as cheese, its future is uncertain (Parsons, 2010). The need to find effective, economic means of dealing with water pollutants associated with agriculture is important.

#### **2.3.2.1** *Cheese-production wastewater*

A specific type of wastewater common in Vermont is from cheese-production facilities. Cheese producing facilities typically generate three types of waste; cheese whey, second cheese whey, and cheese whey wastewater (Carvalho, et al., 2013). Cheese whey is left over from hard cheese production and can be used to create soft cheese products like cottage or curd cheese. Second cheese whey is the by-product of cottage cheese production. Cheese whey wastewater (CWW) is the mixture of the residues left over from both hard- and soft-cheese production. CWW is generally composed of lactose, proteins, and alkaline compounds (cleaning agents, some of which can contain phosphorus) ("Method and composition for cleaning dairy equipment," 1978, "The Chemistry of Cleaning – Essential Industries," n.d., "Types Of Cleaning Detergents," n.d.). CWW pH can range from 4.0 – 13.5, but is generally acidic, a characteristic trait of whey in general. Alkalinity is due to cleaning agents (Danalewich, et al., 1998). CWW is chemically

characterized by elevated organic matter (0.8 – 77 COD, 0.6 – 16 g/L BOD), high salinity due primarily to calcium and sodium salts (Carvalho et al., 2013; Danalewich et al., 1998; Janczukowicz, et al., 2007). TSS can be very high, ranging from 100 – 5000 mg/L. TP values typically are found in the 6 – 280 mg/L range (Carvalho et al., 2013).

Typical treatment methods for CWW include anaerobic digestion, aerobic digestion, physico-chemical processes such as coagulation/flocculation, precipitation, and oxidation (Carvalho et al., 2013; Janczukowicz et al., 2007). Constructed wetlands have more recently been recommended for treatment as well (Carvalho et al., 2013; Comino, et al., 2011). While CWW treatment costs vary widely, and depend on both treatment process and regulatory targets, a study of cheese and food producers in Wisconsin in 2012 found that reducing effluent TP from 2.0 mg/L to 1.0 mg/L would cost all producers in the State approximately \$440 million (Umhoefer, 2012).

In Vermont, cheese producers either send their wastewater to a municipal WWTF, can operate their own on-site treatment plants, or use a combination of the two. Discharging to a municipal WWTF requires a discharge permit from the VT Agency of Natural Resources under the Vermont Water Pollution Control Permit Regulations, based on Section 40 CFR Part 403, which is part of the U.S. EPA's National Pretreatment Program forbidding certain industrial producers from discharging to a municipal WWTF without permission and pretreatment where applicable (U.S. EPA, 2011d). This can mean that individual producers of cheese must build and operate a pre-treatment facility, often at significant expense to themselves.

In Swanton, the cheese production facility in this study had been looking for a lower-cost solution to treat its effluent. The costs associated with paying the municipal WWTF to treat the facility's wastewater, particularly with respect to P were over \$5,000/month as stated by James Irish, superintendent of operations at the Swanton municipal WWTF in 2011 (Irish, pers. comm., 2011). This arrangement had proven to be more expensive than originally anticipated by the cheese production facility owners according to Reg Belliveau, Town of Swanton Manager in 2011 (Belliveau, pers. comm., 2011). For this reason, the cheese production facility owner's, Lotito Foods, Inc., dba Swan Valley Cheese, were looking for a lower-cost option to pre-treat the facility's wash water specifically for P.

### **2.3.3 EAF Steel Slag Filters**

Electric arc furnace (EAF) steel slag filters are filters based on the solids and phosphorus reducing properties of the slag media, a highly-porous aggregate media. Research on this type of material began in the late 1980s and early 1990s as part of larger studies examining the use of alternative substrate media for constructed wetlands for wastewater treatment (Drizo, et al., 1997). The research built on concepts from traditional wastewater engineering using materials rich in Ca, Fe, and Al oxides to enhance P removal (Drizo, et al., 1999).

EAF filters have previously been tested in Vermont to gauge potential to treat dairy farm effluent with applications in both a constructed wetland and in a stand-alone filter (Weber, et al., 2007). Weber et al. (2007) found that EAF steel slag within a constructed wetland

removed up to 75% of Dissolve Reactive Phosphorus (DRP), and as a stand-alone filter removed 72% of DRP. They were further tested as stand-alone filters configured using an ‘in-series’ filter design to treat milk parlor effluent and found that DRP removal efficiency was between 56 – 71% and Total Phosphorus (TP) removal was ~74%. (Bird and Drizo, 2010). Other studies in Vermont focused on treatment of barn yard runoff with EAF steel slag in drainage channels, as well as the effect of operating a filter with high-concentration influent, then allowing the filter to ‘rest’ before putting it back into operation to gauge the potential rejuvenation effect of the resting period (Drizo et al., 2008; Drizo, et al., 2005).

One important finding of these previous studies is that contact time between various steel slags (including EAF) and wastewaters tends to increase the pH of the treated effluent. For EAF steel slags in particular, high effluent pH has been noted, especially in startup and early operation phases of treatment practices (Bird & Drizo, 2010; Lee et al., 2010; Drizo, et al., 2008; Weber et al., 2007). These same studies observed that with increased hydraulic residence time (>24 hours), P removal increased to nearly 100% while pH increased to >11. Recent research by Drizo and Picard (2014), building on Bird & Drizo’s (2010) finding that a hydraulic residence time <24 hours results in lower pH. Additionally, using a pH reducing compartment filled with wood chips, charcoal, or other organic material can further help manage pH increases.

Elevation in pH is thought to be due to the dissolution of Ca oxides present on the slag’s porous surface, which will form OH<sup>-</sup> groups:  $\text{CaO} + \text{H}_2\text{O} \rightarrow \text{Ca}^{2+} + 2\text{OH}^-$ . This interaction

is critical to EAF steel slag's ability to sequester P as the  $\text{Ca}^{2+}$  ions react with phosphates to form a number of Ca-P species depending on solution composition (pH, P concentration, supersaturation index). These species are, generally, amorphous calcium phosphates at lower pH (7-9), which are thought to then convert to octocalcium phosphate in a precursor phase to forming hydroxyapatite ( $\text{Ca}_5(\text{PO}_4)_3$ ).

#### **2.3.4 Research Objectives**

The primary objective of this study was (1) to conduct a field-test of a multi-compartment filter's removal of TP and DRP from cheese production wastewater with innately high P concentrations. Secondary objectives were (2) to test each filter compartment's effectiveness at removing TSS from wastewater and (3) to test the effectiveness of a naturally acidic, organic material-based medium (wood chips) for lowering the pH of wastewater after it had been in contact with EAF steel slag. The results of this study will aid in understanding of how different filter compartments designs and contents influence wastewater treatment performance.

### **2.4 METHODS**

#### **2.4.1 Description of Research Site**

The study was conducted at the Swan Valley Cheese plant's wastewater pre-treatment facility in Swanton, VT. The pre-treatment facility consists of an approximately six-acre (2.42 ha) open aerated lagoon. The facility treats around 20,000 - 40,000 gallons of wastewater daily (rate depends on production at the plant). Once pre-treated, effluent is sent to the Town of Swanton's Municipal Wastewater Treatment Facility. There it undergoes a phosphorus-removal process that relies on precipitation using alum (hydrated

potassium aluminum sulfate). The water is then discharged to the Mississquoi River which drains to Lake Champlain, while the resultant sludge is dried, stored, and subsequently shipped for landfilling.

#### **2.4.2** *Filter Installation and Design*

In December 2011, the filter was constructed near the edge of the Swan Valley Cheese's pre-treatment facility lagoon on a leveled compacted-gravel pad, located approximately 10 vertical feet (3.3 m) uphill from the pre-treatment facility's aerator control and pump room. Wastewater that has been pre-treated in the lagoon is pumped uphill in 1" (2.54 cm) closed-cell foam insulated and heat-taped PVC tubing to the filter inlet. Pumping to the filter only occurs when the pre-treatment facility is discharging to the Municipal WWTF during production. Flow from the lagoon's discharge pipe to the filter is controlled by a Rotameter flow-meter model FL-1500A (Figure 2-1). Prior to entering the flow-meter, water passes through a 3M-brand AquaPure water filter to prevent solids and algae clogging of the sensitive flow-meter.



Figure 2-1: Water filter and flow meter setup to monitor influent rate.

### **2.4.3 Filter Configuration**

The filter described in this study was built based on Drizo & Picard's patent design, which was submitted in 2010 and approved in 2014 (Drizo and Picard, 2014). Based on over 15 research and demonstration field projects led by Drizo at the University of Vermont 2004-2012, it was concluded that EAF steel slag filter should have a hydraulic residence time of  $\sim 24$  h or less in order to ensure that the pH of the final effluent does not increase over 11 so that when modified with pH attenuation units it remains at or below 8.5 (Drizo & Picard, 2014). The filter was designed to receive  $\sim 10$  m<sup>3</sup> wastewater/day (e.g. 7 L/min)

and was constructed accordingly, consisting of 3 compartments (Table 2-1). Table 2-1 describes the filter's compartments. Figure 2-2 shows the multi-compartment filter system in (A) section view and (B) plan view.

The filter system is contained by a pre-cast 6" (15.24cm) thick concrete box manufactured by CAMP Precast of Milton, VT (<http://www.campprecast.com/>). The concrete box was cast in a top and bottom section made watertight by a rubber gasket to seal the inner dividers and outer walls. The filter is divided into three compartments each separated by 4" (10.16cm) thick concrete walls and individually sealed with rubber gaskets to prevent short-circuiting between compartments.

Table 2-1: Filter Compartment Descriptions

Compartment	Volume	Description
1	271 gallons (1.02 m3)	Empty - designed as a solids-settling compartment
2	2046 gallons (7.75 m3)	Filled with EAF steel slag - designed to provide contact time between wastewater and EAF steel slag
3	439 gallons (1.66 m3)	Filled with wood chips (Eastern White Cedar, <i>Thuja occidentalis</i> ) - designed to reduce pH of wastewater



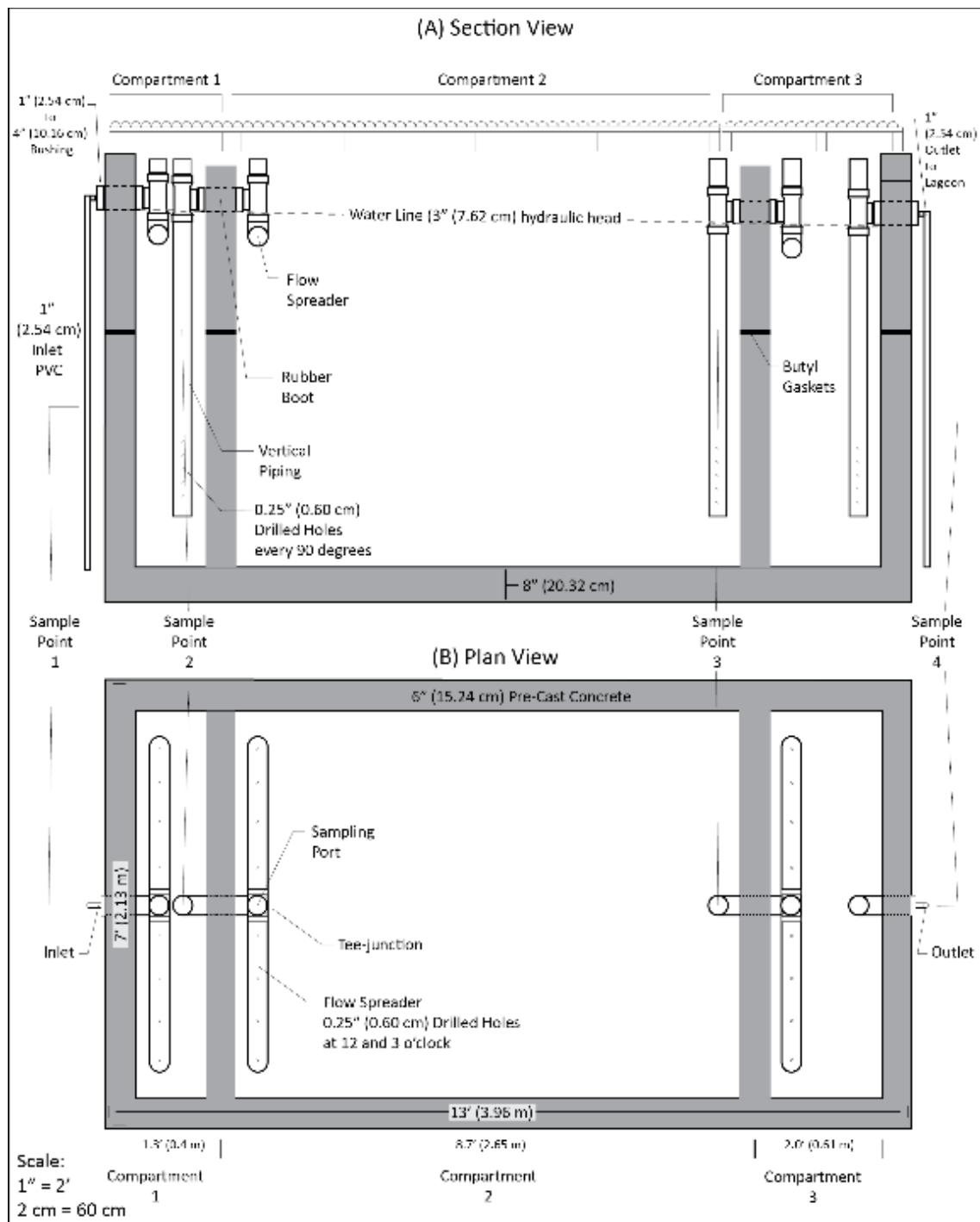


Figure 2-2: Filter schematic. (A) Section view of filter compartments (B) Plan view of filter compartments showing compartment layout and dimensions, wastewater distribution pipes, and sampling points.

The filter's outside dimensions are 13' (3.96 m) L X 7' (2.13 m) W X 7' 8" (2.34 m) H, with a total inner volume of 3,000 gallons (11.35 m<sup>3</sup>) and a media and slag volume of 2,756 gallons as measured to each compartment's pipe outlet. At the inlet, which is 3" (7.62 cm) above the outlet invert elevation, the 1" (2.54 cm) PVC inlet pipe attaches to a rubber gasket set into the concrete that expands to accommodate a 4" (10.16 cm) PVC pipe on the inner side of the wall. A horizontal flow spreader is used at the inlets of each compartment; it consists of a 4" (10.16 cm) PVC tee-junction with two sections of 4" (10.16 cm) PVC pipe with hand-drilled 0.25" (0.60 cm) holes at the top of and 45 degrees from the top on the downstream side of the pipe to allow water to escape. The distal ends of the flow spreader pipes are capped.

The outlets are each constructed of vertical 4" (10.16cm) PVC connected to a 4" (10.16cm) PVC elbow junction that attaches to another rubber gasket through the interior unit walls. 0.25" (0.60 cm) holes are drilled in the lowest 3" (7.62 cm) of vertical piping to allow for infiltration. The bottom of the each vertical outlet pipe is open. This configuration forces a flow path that infiltrates from the top of the filter, down through the filter media and is forced up through the vertical outlet pipe via hydraulic head to prevent short-circuiting (i.e. water flowing directly from inlet pipe to outlet pipe) and to maximize contact time with filter media.

Above the elbow junction at each outlet is a vertical pipe that serves as a sampling port, each 6" (15.62 cm) long and 4" (10.16 cm) diameter. These vertical pipes that serve as sampling ports extend above the tops of inner baffle walls separating each compartment

to prevent short-circuiting via the sampling point pipes. The final outlet pipe is a 1" (2.54 cm) PVC pipe that feeds directly back into the pre-treatment lagoon. No wastewater from the filter was discharged directly to the municipal WWTF.

The entire structure is capped with a flat roof, insulated with 1" (2.54 cm) thick foam insulation and finished with a polycarbonate corrugated waterproof roofing material to guard against freezing in the winter and to protect Compartment 2 from dilution from rain. The roof was constructed in three sections so that the sections covering Compartment 1 and Compartment 3 could be opened independently of Compartment 2. This was to allow for the planting of Compartment 3.

The 271-gallon (1.02 m<sup>3</sup>) Compartment 1 of the filter was left empty. This was intended to provide for solids settling, as with a sediment forebay, and to prevent clogging of the EAF steel slag filter media in Compartment 2. This type of design is similar to most septic tank configurations which feature an initial vault separated from subsequent treatment vaults by a baffle wall. The 2,046 -gallon (7.75 m<sup>3</sup>) Compartment 2 was filled with EAF steel slag obtained from Harsco Metals and Minerals (<http://www.harscomineraux.com/>) located in Contrecoeur Quebec, Canada. EAF steel slag has a porosity of approximately 40%, leaving an approximate pore volume in Compartment 2 of 820 gallons (3.10 m<sup>3</sup>). The final 439-gallon (1.66 m<sup>3</sup>) Compartment 3 was filled with wood chips (Eastern White Cedar, *Thuja occidentalis*) contained in nine wire gabion baskets. The top layer of the unit was planted using transplanted wild *Juncus effusus* (Common Rush) and *Typha latifolia* (Common Cattail) on May 4, 2012 (the insulated winter cover for this section

was removed to allow this).

#### **2.4.4 Sampling Locations**

The filter had four sampling points (See Figure 2-2 and Table 2-2). The Influent sample (Sample Point 1) came directly from a valve-controlled port in the main feed line for the filter, housed in the pre-treatment facility's pump house. The Compartment 1 effluent sample (Sample Point 2) came from the vertical standpipe at the outlet of Compartment 1. The Compartment 2 effluent sample (Sample Point 3) came from the vertical pipe at the outlet of Compartment 2. The Compartment 3 effluent sample (Sample Point 4) came from the filter's outlet pipe (a 1" (2.54cm) pipe attached to the outlet Compartment 3).

Table 2-2: Sampling Point and Filter Stage

Sampling Point	Filter Stage
1	Influent to Filter
2	Effluent of Compartment 1
3	Effluent of Compartment 2
4	Effluent of Compartment 3 (Final Filter Effluent)

#### **2.4.5 Study Period**

The filter was initially put into operation on December 21, 2011 and operated until February 29, 2012. Due to sub-freezing temperatures, filter operation was discontinued until May 4, 2012 when it was put back into operation until October 10, 2012. During the period of inactivity, the filter was not emptied due to a lack of a drain mechanism.

Wastewater was allowed to remain in contact with the contents of the filter (EAF steel slag and wood chips).

#### **2.4.6** *Filter Influent – Operation*

In the configuration described above, the filter is only fed when the pre-treatment facility is actively discharging wastewater to the municipal WWTF. This rate is affected by cheese production volume at the SVC plant, as well as precipitation rates and temperature at the pre-treatment facility's lagoons, which can increase lagoon volume with rain or decrease it with evaporation. The filter was therefore only fed dependent on plant and pre-treatment facility operating conditions. This was not part of the original experimental design but rather a condition that occurred during operation. It is known that extended contact time with EAF steel slag can have a negative effect on the pH of effluent wastewater.

The filter's influent rate was set once weekly using the flow-meter. This influent rate was directly dependent on the pre-treatment facility's discharge rate – if that rate was changed as a result of plant operations, to accommodate any of the factors listed above, the influent rate to the filter would change. The control researchers had over the filter's influent rate was a limitation in this study. There were times when flow to the filter was decreased to where little to no influent was pumped to the filter. When there was no effluent from the aeration lagoons, the volume of water remaining in the filter stayed static, increasing retention time; the filter was never emptied during periods of non-feeding.

There were times during the study period when the filter is assumed to have stagnated,

based on communications with pre-treatment facility staff, but the dates and times of these periods of stagnation are not known. As mentioned previously, extended periods of contact with EAF steel slag could result in elevated pH of wastewater effluent from the filter. Stagnation of wastewater could also result more settling of particles in Compartment 1, as well as growth of algae in Compartments 1 and 3 due to high nutrient concentrations and still water.

#### **2.4.7** *Sample Collection and Analysis*

Samples were collected on a weekly basis using grab-sampling methods for a period of 152 days from May 4 to October 10, 2012 following procedures outlined by the US EPA and supplemented by additional guidance issued by the VT Department of Environmental Conservation (US EPA, 2013; VT DEC, 2006). One sample was collected per compartment and samples were analyzed separately. Samples were not composited. All bottles were triple-rinsed with sample before obtaining the final sample. Filter Influent at Sample Point 1 was sampled using a 1-L Nalgene bottle that had previously been acid-washed and triple-rinsed with reverse osmosis water.

Samples from Compartments 1 and 2 were obtained using a 250 ml Nalgene bottle attached to a 3' (0.91m) pole so as to be able to insert the bottle into the vertical effluent pipes and sample from the middle of the water column (Figure 2-2). The Compartment 3 effluent sample came was collected directly from the outlet pipe using a 1-L bottle (Table 2-2).

Samples were analyzed for total phosphorus (TP), dissolved reactive phosphorus (DRP),

total suspended solids (TSS), and pH. Samples to be analyzed for TP, DRP, and TSS were transported from the site in Swanton back to the University of Vermont's Agriculture and Environmental Testing Laboratory using insulated coolers and ice packs to maintain samples at 4° C for the 45-minute trip. At the laboratory, 15 ml of each sample was filtered for DRP analysis using a 33mm diameter Fisher Scientific brand 45µm nylon syringe filter and frozen for batch analysis. 50 ml of each sample was separated unfiltered and refrigerated at 4 degrees Celsius for digestion for TP (APHA Method 4500-P B).

TP concentrations were determined by taking 5 mL of previously stored samples and performing a potassium persulfate digestion to convert particulate P into DRP. Samples were analyzed using the same instrument as for DRP following digestion (APHA Method 4500-P B).

DRP concentrations were determined by taking previously filtered and frozen samples out of the freezer and allowing them to thaw completely. Analysis was conducted using a Lachat Instruments Flow Injection Analysis spectrometer to determine DRP concentrations using a standard molybdate color reaction with ascorbic acid (APHA Method 4500-P B).

TSS samples were refrigerated for bulk analysis. TSS concentrations were determined gravimetrically by first weighing each 45µm filter, filtering up to 1 L or until filter clogged, of wastewater through the filter using vacuum suction and allowing the filter to dry for 24 hours at 103 degrees Fahrenheit, then re-weighing the filter to determine weight of sediment trapped and correcting for volume (APHA Method 2540 D).

Analysis of pH was performed in the field at the pre-treatment facility's aerator control and pump room using a calibrated Hach HQ11d with PHC201 pH electrode.

#### **2.4.8** *Statistical Analysis*

The entire set of data was analyzed using JMP®, Version 12.1. SAS Institute Inc., Cary, NC, 1989-2007. Outlying values were explored using quantile outliers analysis. Once any outlying values were identified, distributions for all datasets were generated. Differences between compartment analyte concentrations were determined using a nonparametric Wilcoxon method comparison of median values and the cumulative distribution function. Correlation was analyzed using the fit X by Y function within JMP. JMP and Microsoft Excel were used to generate figures and charts



## **2.5 RESULTS**

### **2.5.1** *Total Phosphorus (TP), Dissolved Reactive Phosphorus (DRP), and Total Suspended Solids (TSS) overall removal efficiencies during filter operation*

Table 2-3: Nonparametric Wilcoxon method comparison for all analytes across all filter compartments showing statistical significance (*p*-value), sample number (*n*), and the difference between median values. The last line denotes the overall change from initial filter influent to final filter effluent. \* denotes statistical significance.

Analyte	Comparison	<i>p</i>	Median Difference (mg/L)
TP	<i>Influent - Compartment 1 Effluent</i>	0.830	1.80
	<i>Compartment 1 Effluent - Compartment 2 Effluent</i>	<0.0001*	-25.06
	<i>Compartment 2 Effluent - Compartment 3 Effluent</i>	0.145	0.26
	<i>Influent - Compartment 3 Effluent</i>	<0.0001*	-23.00
DRP	<i>Influent - Compartment 1 Effluent</i>	0.406	-1.34
	<i>Compartment 1 Effluent - Compartment 2 Effluent</i>	<0.0001*	-18.01
	<i>Compartment 2 Effluent - Compartment 3 Effluent</i>	<0.0001*	0.48
	<i>Influent - Compartment 3 Effluent</i>	<0.0001*	-18.87
TSS	<i>Influent - Compartment 1 Effluent</i>	0.239	18.24
	<i>Compartment 1 Effluent - Compartment 2 Effluent</i>	0.0171*	-39.45
	<i>Compartment 2 Effluent - Compartment 3 Effluent</i>	0.577	-6.91
	<i>Influent - Compartment 3 Effluent</i>	0.0167*	-28.12
Analyte	Comparison	<i>p</i>	Median Difference (level)
pH	<i>Influent - Compartment 1 Effluent</i>	0.029	0.26
	<i>Compartment 1 Effluent - Compartment 2 Effluent</i>	0.005*	3.04
	<i>Compartment 2 Effluent - Compartment 3 Effluent</i>	0.0002*	1.05
	<i>Influent - Compartment 3 Effluent</i>	0.0008*	2.25

Table 2-4: Summary of median, n, and Standard Deviation (SD) for all samples taken during filter operation for each sample point.

Analyte	Point	Median (mg/L)	n	SD (mg/L)
TP	Influent	24.00	23	6.69
	Compartment 1 Effluent	25.80	16	7.80
	Compartment 2 Effluent	0.74	21	0.91
	Compartment 3 Effluent	1.00	23	2.54
DRP	Influent	19.52	21	5.06
	Compartment 1 Effluent	18.18	18	4.18
	Compartment 2 Effluent	0.17	21	0.19
	Compartment 3 Effluent	0.65	21	5.04
TSS	Influent	53.81	22	44.54
	Compartment 1 Effluent	72.05	13	89.06
	Compartment 2 Effluent	32.60	17	36.58
	Compartment 3 Effluent	25.69	22	49.28
Analyte	Point	Median (level)	n	SD (level)
pH	Influent	7.87	15	0.25
	Compartment 1 Effluent	8.13	7	0.25
	Compartment 2 Effluent	11.17	12	0.40
	Compartment 3 Effluent	10.12	15	1.55

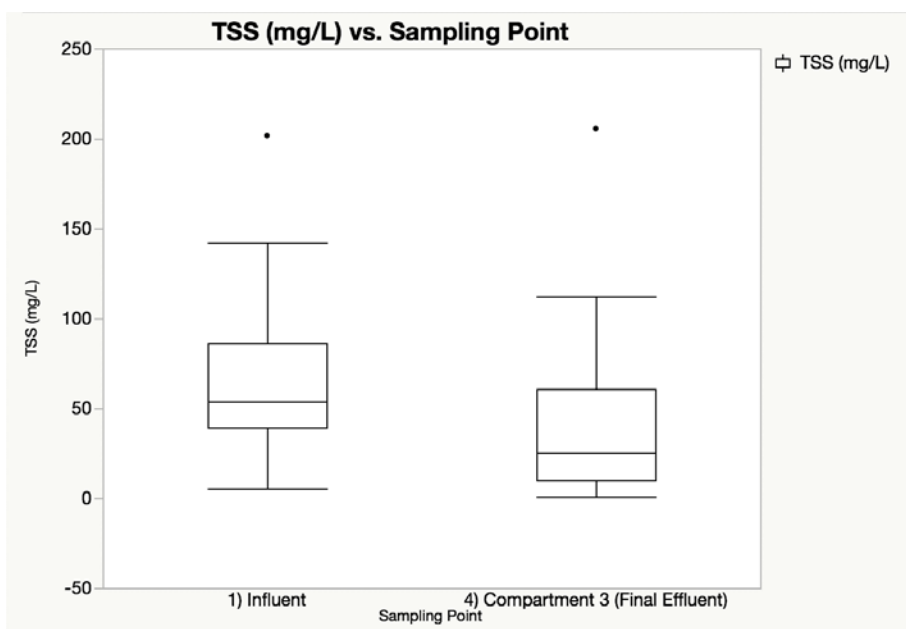
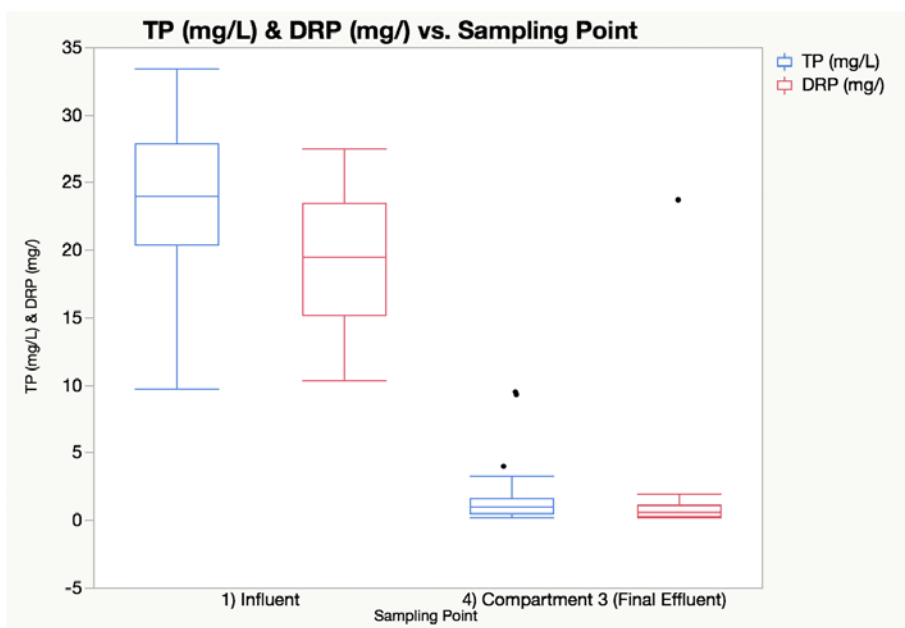


Figure 2-3: TP, DRP, and TSS influent and effluent concentrations during 152 days of filter operation from May 4 to October 10, 2012. Influent is taken at Sample Point 1 (prior to entering Compartment 1) while effluent is taken at Sample Point 4 (after the outlet of Compartment 3).

Removal rates for TP, DRP, and TSS median concentrations from influent to effluent (i.e. measuring the whole filter system's performance) during the study period were 95.83%, 96.65%, and 52.25% respectively. Influent to effluent median concentrations differences were found to be statistically significant for all analytes (See Table 2-3).

### 2.5.2 Total Phosphorus Inter-Compartmental Results

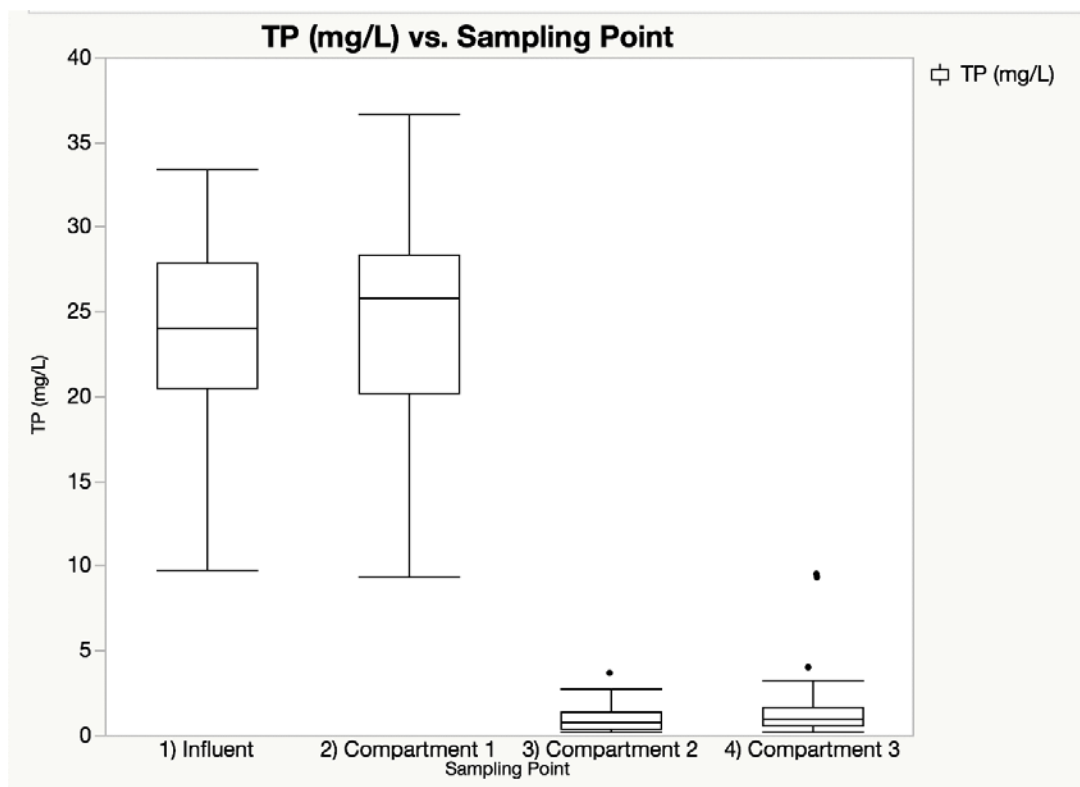


Figure 2-4: Inter-compartmental difference between TP concentrations.

Median Compartment 1 effluent to Compartment 2 effluent (EAF slag filter effluent) TP-concentrations demonstrated a decrease of 97.12% from 25.80 mg/L  $\pm$  7.80 mg/L to 0.74 mg/L  $\pm$  0.91 mg/L (Figure 2-4, Table 2-4). This difference was statistically significant

(Table 2-3). No other statistically significant differences were found between compartments.

### 2.5.3 Dissolved Reactive Phosphorus Inter-Compartmental Results

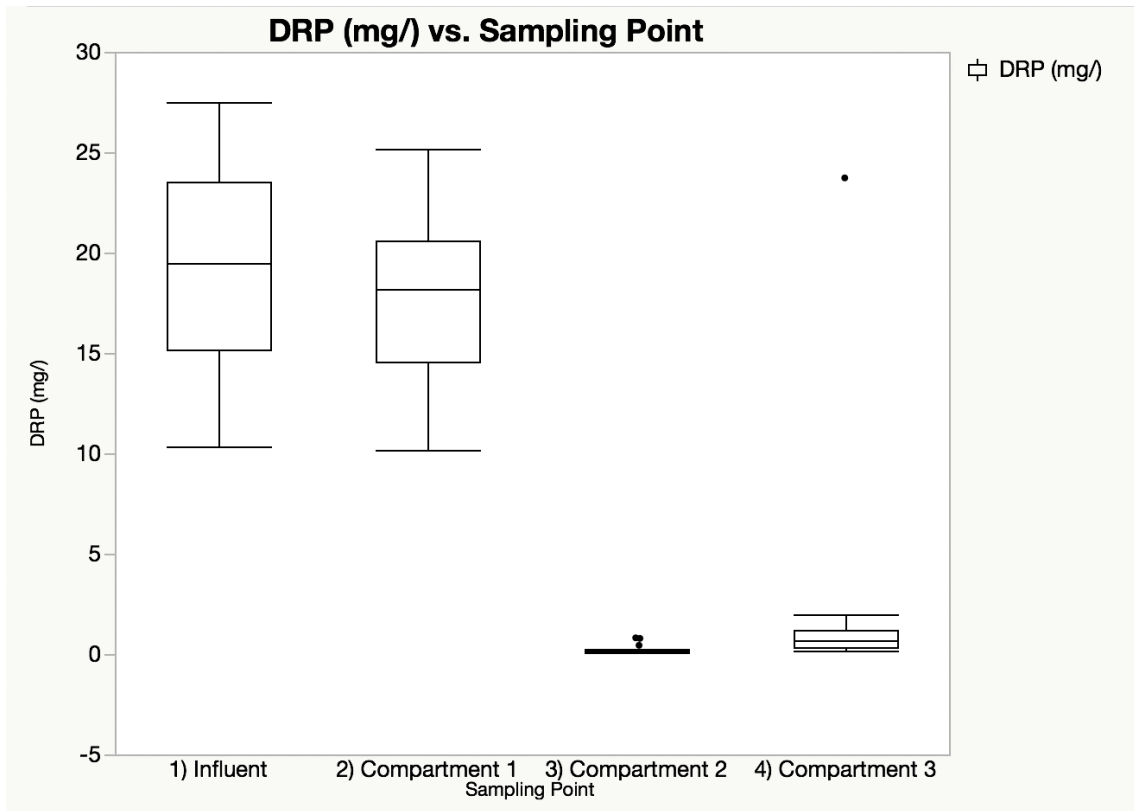


Figure 2-5: Inter-compartmental difference in DRP concentrations.

DRP concentrations from Compartment 1 effluent to Compartment 2 effluent were marked by a 99.05% decrease from 18.18 mg/L  $\pm$  4.18 mg/L to 0.17 mg/L  $\pm$  0.19 mg/L (Figure 2-5, Table 2-4). From Compartment 2 effluent to Compartment 3 effluent, concentrations increased 279.06% from 0.17 mg/L  $\pm$  0.19 mg/L to 0.65 mg/L  $\pm$  5.04 mg/L (Figure 2-5, Table 2-4). These differences were statistically significant (Table 2-3). No other statistically significant differences were found between compartments.

#### 2.5.4 Total Suspended Solids Inter-Compartmental Results

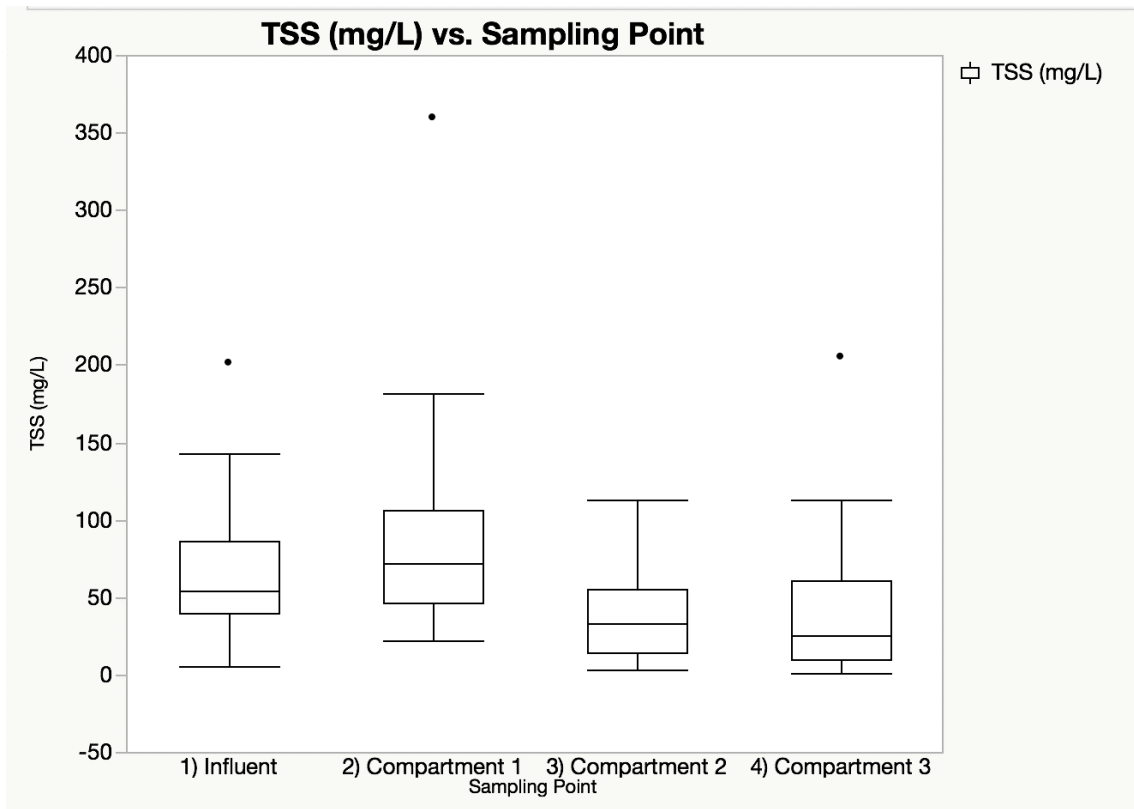


Figure 2-6: Inter-compartmental difference in TSS concentrations.

Between Compartment 1 effluent and Compartment 2 effluent, TSS decreased 54.75% from 72.05 mg/L  $\pm$  89.06 mg/L to 32.60 mg/L  $\pm$  36.58 mg/L (Figure 2-6, Table 2-4). This difference was statistically significant (Table 2-3). No other statistically significant differences were found between filter compartments.



### 2.5.5 pH Results

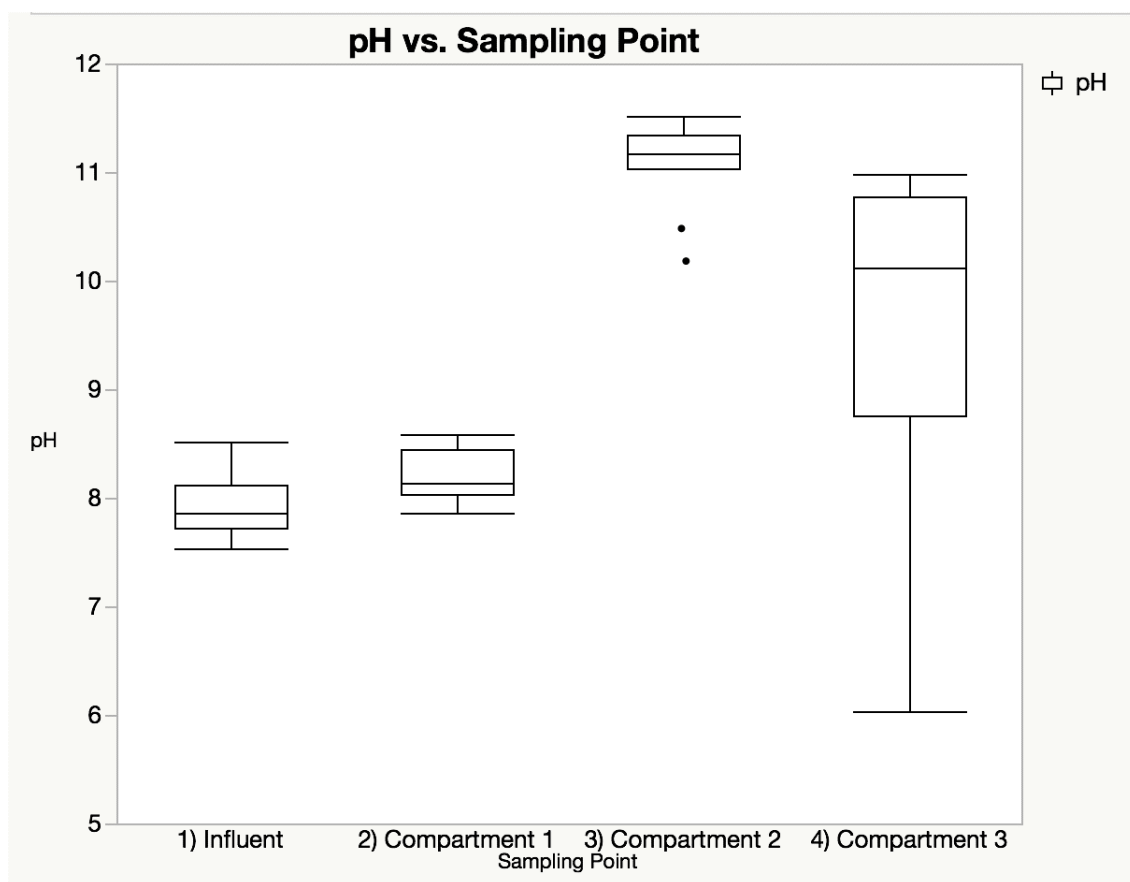


Figure 2-7: Inter-compartmental differences in pH levels.

The median pH for influent was  $7.87 \pm 0.25$  ( $n = 15$ ) while the average for Compartment 3 effluent pH was  $10.12 \pm 1.55$  ( $n = 15$ ) (Figure 2-7, Table 2-4). Between compartments, pH varied. From the influent to the Compartment 1 effluent the median pH increased from  $7.87 \pm 0.25$  to  $8.13 \pm 0.25$  (Figure 2-7, Table 2-4). From the Compartment 1 effluent to the Compartment 2 effluent, pH again increased from  $8.13 \pm 0.25$  to  $11.17 \pm 0.39$  (Figure 2-7, Table 2-4). Finally, from the Compartment 2 effluent to the Compartment 3 effluent, pH decreased from  $11.17 \pm 0.39$  to  $10.12 \pm 1.55$  (Figure 2-7, Table 2-4).

The differences between pH levels were found to be statistically significantly different between Compartment 1 Effluent and Compartment 2 Effluent, between Compartment 2 Effluent and Compartment 3 Effluent, and between Influent and Compartment 3 (Final) Effluent (Table 2-3). No statistically significant difference was found between Influent and Compartment 1 Effluent.

#### **2.5.6** *Comparison of TP/DRP to pH*

It was observed that TP and DRP were statistically significantly different from influent to effluent (Table 2-3) (effluent was lower than influent for both analytes). It was likewise observed that pH was statistically significantly different from influent to effluent (effluent was higher than influent) (Table 2-3). A regression comparison of these two factors (phosphorus concentration and pH level) was made to investigate the relationship between the two (Figure 2-8).

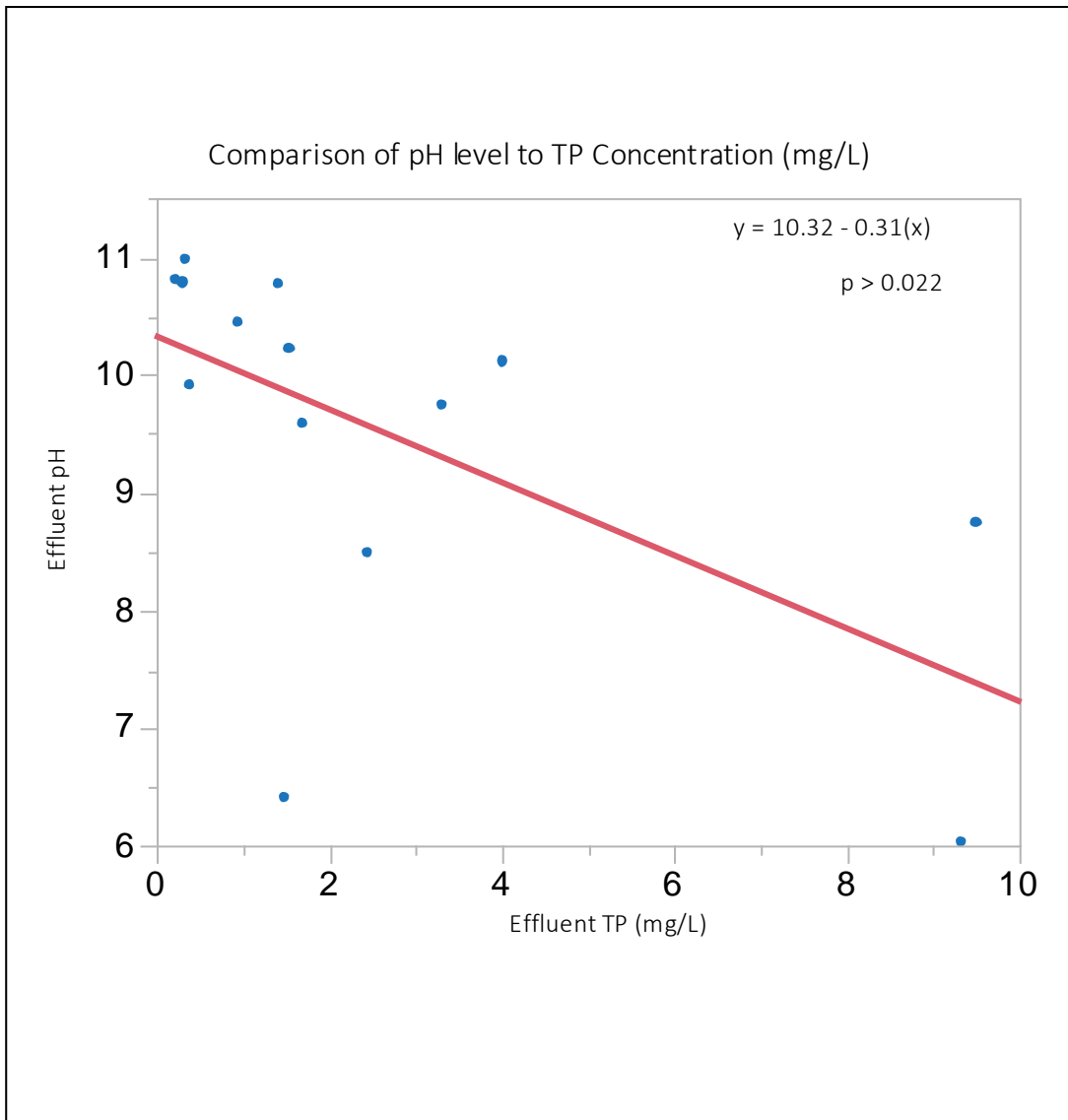


Figure 2-8: Linear regression comparison of pH level to TP concentration (mg/L)

For TP compared to pH, using a bivariate analysis of two continuous variables for  $n = 14$ , pH level corresponded negatively with TP concentration – a decrease in TP concentration corresponded to an increase in pH level with  $p > 0.022$ . This value indicates a statistically significant relationship between increases in pH and TP concentration decreases. The

same analysis was performed for DRP versus pH where  $n = 11$  (Figure 2-9). It was also found that pH corresponded negatively with DRP concentration, where the correlation was statistically significant as  $p < 0.001$ .

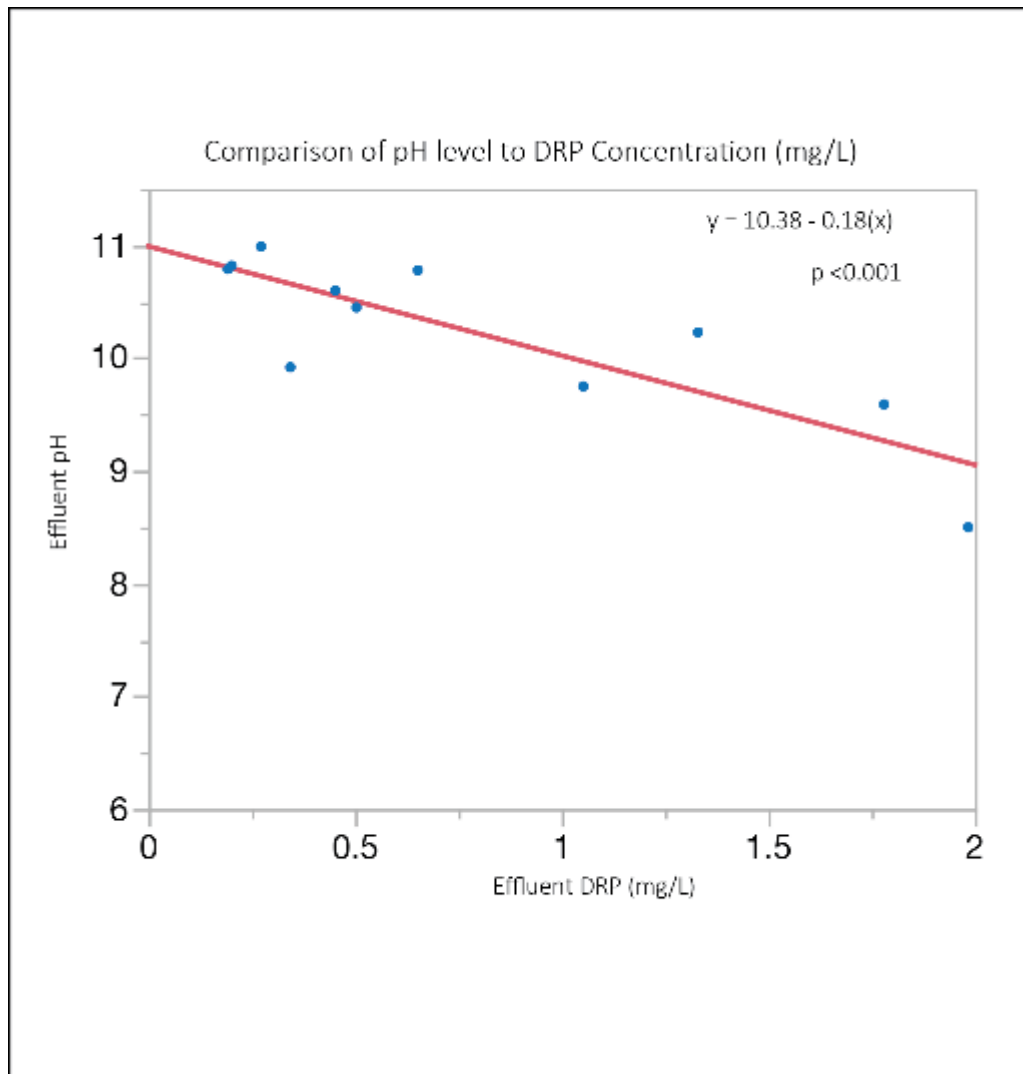


Figure 2-9: Linear regression comparison of pH with DRP concentration (mg/L)

## DISCUSSION

### **2.5.7** *Total Phosphorus Removal Rates During Filter Operation*

As shown in Table 2-4, the median influent TP concentration during 152 days of filter operation was found to be 24.00 mg/L  $\pm$  6.69 mg/L for a total of 23 separate grab samples, while median effluent TP concentration was found to be 1.00 mg/L  $\pm$  2.54 mg/L. This represents an average reduction in median TP concentration of 95.83% over the study period.

### **2.5.8** *Dissolved Reactive Phosphorus Removal Rates During Filter Operation*

As shown in Figure 2-5 and Table 2-4, DRP concentrations for both the influent and the Compartment 1 effluent were relatively high compared to Compartment 2 and Compartment 3 effluent. DRP concentration showed no statistically significant difference between filter influent and Compartment 1 effluent, displaying a slight increase of 6.8%. This is not surprising as Compartment 1 was designed only for physical solids settling, which would not affect nutrients in solution such as DRP. As discussed previously, P-removal is governed by adsorption, precipitation, and physical filtration. The only substance with which DRP could interact chemically in Compartment 1 were the concrete walls forming the compartment. Research has shown that concrete, even in a more porous crushed form, does not have a significant effect on P concentration (Li et al., 2016).

Between Compartment 1 and Compartment 2 effluent a statistically significant reduction in DRP occurred, indicating good performance by the EAF steel slag filter, while between Compartment 2 and Compartment 3 effluent was shown to have a small increase in DRP

concentration (Figure 2-5 and Table 2-3), which was counterproductive after the removal of DRP by Compartment 2. The Compartment 3 contained acidic Eastern White Cedar (*Thuja occidentalis*) wood chips meant to neutralize effluent from Compartment 2, made more alkaline by contact with EAF steel slag. As wood chips themselves contain P, it is probable that some DRP may have been leached out of this material and been measured in the effluent DRP concentrations. Research has found DRP leaching from woody debris in runoff in Auerswald & Weigand, (1996), as well as in woodchip bioreactors (saturated anaerobic treatment structures primarily focused on nitrogen removal from agricultural runoff) in Sharrer, et al., (2016). In a review of studies conducted by Sharrer, et al. (2016), it was found that these woodchip bioreactors could act as either a sink or source of DRP, depending on hydraulic residence time, woodchip material, and wastewater influent type. Specific concentrations were frequently not reported for these studies; however, in a similar study, Healy, et al. (2015) found that DRP concentrations from woodchip bioreactors could reach up to 1.1 mg/L. Additionally, Compartment 3 was planted with *Juncus effusus* and *Typha latifolia*, both rhizomatous macrophytes with extensive root systems. Plant root decomposition may have contributed to this rise in DRP concentration, though concentrations measured at the outlet of Compartment 3 were still less than 1.0 mg/L DRP – much lower than the filter influent.

#### **2.5.9 Mechanisms of Phosphorus Removal – TP and DRP**

Previous studies have found that the P-removal performance using EAF steel slag depends primarily on chemical adsorption and precipitation of P to Ca and Fe species that comprise the majority of the slag's chemical composition, as well as physical filtration of

particulate P via collection of suspended solids in the slag's porous structure which is a noted removal mechanism in other media types as well (Yamada, et al., 1986; Zhu, et al., 1997). EAF steel slag typically has higher Ca (~40%) than Fe (35%) (Drizo et al., 2002, 2006). In addition to Ca, Fe also plays a role in P-removal by slag as both Fe and Ca are known to bind with P (Sims & Pierzynski, 2005). In a study of constructed wetlands using EAF steel slag as the primary substrate, Barca, et al., (2013) found that P-removal was accomplished primarily by Ca-O slag dissolution, followed by Ca-P precipitation. This is further substantiated by Lee, et al., (2010) and Drizo, et al., (2006) in their research on EAF steel slag filters which found that Ca-O dissolution on slag surfaces provided an adsorption site for phosphate ( $\text{PO}_4^{3-}$ ), typically leading to the formation of hydroxyapatite (HAP), following formation of amorphous calcium phosphates (ACPs) at lower pH ranges. HAP formation typically occurs in alkaline conditions ( $\text{pH} > 9.3$ ) which is typical of wastewaters that come into contact with steel slags in excess of 12 hours (Drizo et al., 2006; Weber et al., 2007). Further, Lee, et al., (2010) found that increased residence times ( $> 24$  hours) led to P removal rates of nearly 100%, similar to the removal rates seen in the current study, along with elevated pH ( $> 11$ ), indicating that HAP formation is the primary driver for P removal at longer residence times. Though careful control of flow rates to the filter were not possible due to the variations in operation of the cheese production facility, it is known that the plant often only operated 1-2 days per week. During periods of inactivity the pre-treatment facility would not discharge wastewater to the municipal WWTF and accordingly no flow was going to the EAF steel slag filter. These 5-6 day periods of stagnation represent a lengthy residence time that would result



in the high P removal rates observed during this study, similar to Lee, et al., (2010).

It is important to note the differences in TP concentration between filter compartments as shown in Figure 2-4 and Table 2-4. Compartment 1, essentially a narrow, tall empty concrete box, showed little to no discernible effect on TP concentration; the increase of 7.5% was not statistically significant and it can be concluded that this compartment had little overall effect on P removal within the filter (Table 2-3). From Compartment 2 effluent to Compartment 3 effluent, TP was shown to increase in median concentration by 34.77% from 0.74 to 1.00 mg/L (Table 2-4) but this was not statistically significant (Table 2-3). However, comparing Compartment 1 to Compartment 2 effluent wastewater showed a statistically significant decrease in concentration by 97.12% from 25.80 to 0.74 mg/L. As Compartment 2 contained solely EAF slag and wastewater was forced to move through the slag matrix via horizontal flow, maximizing potential contact time through inlet and outlet design, this result indicates that EAF steel slag was the primary driver for TP removal from wastewater, even under variable conditions with respect to exposure time of wastewater to EAF slag.

Numerous studies have shown that EAF steel slags have this potential. Drizo et al., (2002) reported that over the course of a 4-day column experiment, EAF steel slag removed up to 86% of P from a synthetic wastewater solution. Studies on EAF steel slag treating dairy parlor effluent found similarly good removal rates for P. Weber, et al., (2007) found a system of stand-alone EAF steel slag filters treating dairy parlor and barnyard runoff waste both with and without pre-treatment by a constructed wetland was able to remove

between 72-75% of DRP over either a 12 or 24 hour residence time. Lee, et al., (2010) conducted research on the same effluent source at the University of Vermont Miller Farm Dairy Operation and compared the percentage removal of constructed wetland effluents with and without an EAF steel slag filter. For constructed wetlands with an EAF steel slag filter, Lee found DRP removal rates in excess of 99%, whereas constructed wetlands without an EAF steel slag filter had lower DRP removal rates between 50–75%. Total hydraulic residence times for these systems varied between ~2.5 and ~5 days over the course of the study period. These lengthy hydraulic residence times are proposed as the primary driver for P removal in Lee's study and may indicate why P removal rates are similarly high in this study. Finally, at this same facility, Bird & Drizo (2010) found that, by using a hydraulic residence time of 18 hours, an EAF steel slag filter was able to remove between 42-76% of DRP from milk parlor and dairy barnyard wastewater, despite repeated issues with solids clogging.

Shilton et al., (2005) found that for EAF steel slag treating pond effluent, removal for real wastewater was actually higher than for synthetic wastewater and that the P-removal rate did not exhibit linear variation with residence time. Drizo et al., (2006) found that filter clogging increased P-removal from 80% to 100% in a column study, though this was attributed to the temporary increase in hydraulic residence time provided by clogging. Chazarenc et al., (2007) found a 75% P-removal rate in a similar column study, further concluding that that rate was dependent largely on the slow release of Ca ions, along with OH<sup>-</sup> ions, which in turn formed hydroxyapatite crystals on the slag surface. The finding that, for a wastewater stream with a high presence of Ca ions and high P concentrations

as seen in this study, >95% of TP would be removed from the influent to Compartment 2 effluent, seems well-supported by previous studies.

#### **2.5.10** *The Effects of Multi-Compartment Layout on Filter Effectiveness with Respect to TSS*

##### **2.5.10.1** *TSS: Influent to Compartment 1*

The filter's multi-compartment layout was an important part of the design to attempt to eliminate solids clogging. Clogging is a noted problem in filter operation generally stemming from either chemical or physical clogging, as well as biological clogging via the formation of biofilms on filter media (Grace, et al., 2016; Hatt, et al., 2008; Johansson-Westholm, 2010; Vohla, et al., 2011). These same issues have been noted for EAF steel slag filters as well (Bird & Drizo, 2010; Drizo et al., 2006; Weber et al., 2007). This has prompted some researchers to suggest the use of specific designs to remove particulates from the wastewater stream prior to water reaching the P-removing filter to avoid clogging and prolong filter life (Chazarenc et al., 2007).

In order to slow clogging in the filter, Compartment 1 was incorporated into the filter design to allow for solids settling. The results between influent and Compartment 1 do not support the conclusion that solids settling occurred as TSS showed an increase of 33.89%, from 53.81 mg/L  $\pm$ 44.54mg/L to 72.05 mg/L  $\pm$ 89.06 mg/ (See Figure 2-4 and Table 2-4).

The data for TSS removal performance by Compartment 1 are highly variable and the reasons for this are unclear. The compartment was intended to provide for solids settling but it is possible that the variable flow rate through the filter resulted in particle

resuspension. However, flow rate data to support this are lacking. Ideally, flow rate information would have been collected reliably but the experimental setup did not allow for this. Collecting effluent rate could have potentially served to provide this information. Future experiments of this type should ensure that accurate flow data is collected. Though each compartment's inlet and outlet configurations were design to initially spread flow horizontally and then force it upward through the vertical outlet pipe (See Figure 2-2), there may have been adequate flow rate in this empty compartment to allow flow to scour and re-suspend solids. This would suggest that the compartment's design (inlet configuration, dimensions, lack of flow-dissipating media) could be responsible for the lack of TSS reduction.

Poor TSS removal could also have been attributable to algal-growth being measured as solids. Algae growth within Compartment 1 may also have resulted in increased TSS between influent and Compartment 1 effluent. For much of the year, the compartment was covered with a roof to prevent freezing or dilution of wastewater by precipitation. Researchers permanently removed the roof sections from Compartments 1 and 3 after putting the filter back into operation on May 4, 2012 to allow for planting of Compartment 3 and to facilitate sampling of Compartment 1. This allowed sunlight to reach wastewater in Compartment 1 at all times. In a nutrient-rich environment, this likely resulted in algae growth, which would have been counted as suspended solids. Additionally, Influent samples were collected prior to the water filter installed to prevent clogging of the flow-meter. This may have lowered Influent TSS levels by filtering out some solids. Combined with internal loading within Compartment 1 due to algae growth, this could possibly

explain the increase in TSS from Influent to Compartment 1 Effluent.

It is also possible that data collection methods were responsible for the TSS levels. Samples were obtained from the mid-point of the vertical pipe at the outlet of Compartment 1 as described in the methods section. Samples were obtained from the middle of the water column as it was thought that this would be the most representative sample of the effluent; samples from lower in the water column could have had higher concentrations of suspended solids while samples from higher in the water column could have had floatables like algae or other bio-solids that would have skewed the data. A better method would potentially involve sampling from the entire length of the pipe and compositing samples to obtain an average effluent concentration.

If the lack of reduction seen from influent to Compartment 1 effluent was indeed caused by either poor design or algae growth, this would indicate the need to implement a more robust solids filter for pre-treatment to prevent clogging of the EAF steel slag. Metcalf & Eddy (2003) note eight different methods for suspended solids removal from wastewater treatment trains: screening, grit removal, sedimentation, high-rate clarification, floatation, chemical precipitation, depth filtration, and surface filtration. Each of these processes has its considerations. Using screening where solids are increasingly filtered out using screens of decreasing mesh pore sizes will result in the need for frequent screened cleaning or replacement. Using sedimentation incorporating baffles and skimmers will result in the need to clean sumps behind baffles and can allow for material resuspension with certain designs. Depth filtration methods using granular-media,

typically sand or anthracite, though other media such as granular activated carbon can be used as well, creates the need to create backflush mechanisms to remove accumulated solids. The method used will depend on a thorough analysis of the types of solids to be filtered and the rate at which influent will reach the pre-treatment filter. This is an area of this particular filter's design that will need future research.

#### **2.5.10.2**      *TSS: Compartment 1 effluent to Compartment 2 effluent*

Effluent from Compartment 2 was found to have a median value of 32.60 mg/L  $\pm$ 36.58 mg/L TSS, a 54.75% decrease from Compartment 1 effluent. This result is statistically significant. This is likely due to two factors – physical filtration of particles suspended in wastewaters by the highly porous EAF steel slag filter media and chemical adsorption of P-rich particles at adsorption sites on EAF steel slag particles (Barca et al., 2012; Bird & Drizo, 2010; Drizo et al., 2005; Shilton et al., 2006; Weber et al., 2007).

Physical filtration of particles by EAF steel slag filter media is primarily dictated by the physical structure of steel slag – it has a porosity of approximately 44% with a high surface-area-to-volume ratio, which has been shown to readily filter particles (Yamada et al., 1986; Zhu et al., 1997).

TSS concentration reduction due to chemical adsorption and physical particle filtration is supported by the results of TP reduction from Compartment 1 effluent to Compartment 2 effluent – TP concentration decreased 97.12% from 25.80 mg/L  $\pm$ 7.80 mg/L to 0.74 mg/L  $\pm$ 0.91 mg/L.

#### **2.5.10.3**      *TSS: Compartment 2 to Compartment 3 effluent (Final effluent)*

Compartment 3, composed of gabion baskets filled with wood chips, as well as the root systems of both *Juncus effusus* and *Typha latifolia*, had no statistically significant effect on TSS concentration, reducing the overall concentration from 32.60 mg/L  $\pm$  36.58 mg/L to 25.69 mg/L  $\pm$  49.28 mg/L, a reduction of 21.19%. This reduction is likely attributable to the physical filtration of suspended particles through the pore spaces between wood chips.

The data support the conclusion that of all the filter compartments, Compartment 2, with its EAF steel slag substrate, had a statistically significant effect on TSS concentration, with Compartment 1 and Compartment 3 producing either an increase in TSS concentration (in the case of the former) or a slight decrease (in the case of the latter). This indicates the future need for research into more effective pre-treatment filtering strategies such as screening, more efficient sedimentation tanks, or depth filters to remove solids and potentially prevent clogging in the EAF steel slag substrate. These results correlate with the results found for TP.

#### **2.5.11** *Filter Operation and pH levels*

As noted previously, the influent rate to the filter was highly variable. For the majority of the research period, the filter was receiving flow as few as only 1-2 days per week, with the remaining 5-6 days being periods of inactivity during which time wastewater was stagnant in the filter according to James Irish, superintendent of the Swanton municipal WWTF in 2011 (Irish, pers. comm., 2011). These long periods of stagnation, which had a demonstrated positive effect on P removal, served to increase the pH of the filter's

effluent. Figures 2-8 and 2-9 illustrate there is statistically significant inverse linear relationship between P concentration (both TP and DRP) and pH where pH increases as P concentration decreases.

While filter influent was essentially unchanged by Compartment 1, Compartment 2 significantly increased pH to an alkaline average of  $11.17 \pm 0.40$  as shown in Figure 2-7. This exact trend has been noted in previous studies of EAF steel slag filters (Lee, et al., 2010, Drizo, et al., 2008). The filter's original design with a relatively large EAF steel slag compartment and a proportionately small pH reducing unit was intended to treat continuously flowing wastewater in an 18-24 hour period, which Bird & Drizo (2010) had found to be the optimal time frame for P removal without excessively elevating pH. Maintaining the design flow rate for the optimal hydraulic residence time was not possible given conditions present during this study and impacted the ability of Compartment 3 to fulfill its function with respect to pH reduction.

It is believed that the periods of stagnation, coupled with the proportionately small volume of wood chips in Compartment 3 resulted in elevated pH. It is also possible that because the wood chips were encased in wire gabion baskets to prevent floatation and placed in the concrete compartment, that small interstitial spaces may have allowed for preferential flows between the baskets and the compartment walls, as well as between the baskets themselves. This would have reduced contact between wastewater and wood chips, decreasing the opportunity for pH reduction. While these preferential flows were not specifically studied, it remains a possibility that they occurred. However,



Compartment 3 was designed and sized to receive continuous flow, in which case it would have decreased pH. Future designs with similar objectives should account for flow variability from the plant and include a large pH attenuation unit.

The median effluent pH of the filter was 10.12, whereas the Vermont Water Quality Standards (2014) dictate that pH discharges from wastewater treatment facilities must fall between 6.5 – 8.5. Effluent from the filter would require further treatment to reduce the pH (Vermont Agency of Natural Resources, 2014).

pH adjustment is a common practice in wastewater treatment (Metcalf and Eddy, 2003). For alkaline wastewater streams, adjustment is most commonly accomplished via acidification using sulfuric, hydrochloric, or carbonic acids (Metcalf and Eddy, 2003). These techniques rely on carefully controlled flows to determine dosing. In the event that carefully-controlled flows can't be achieved for a filter using EAF steel slag to remove P, using one of these techniques may prove necessary to reduce pH to legal discharge levels.

#### **2.5.11.1**      *Flow Rate Variability During Study Period*

As noted in the Methods section, influent flow rate was intended to be controlled by the Rotatmeter flow meter installed on the influent feeding line tapped into Swan Valley Cheese's pre-treatment wastewater treatment plant effluent line. Researchers in partnership with plant operators, attempted control of flow rate at Swan Valley Cheese's pre-treatment facility. Researchers would set the rate weekly, based on flow conditions at the time of a sampling visit to ensure that the filter would continuously receive

wastewater at a relatively constant rate of 7 L/min as per the original design. However, since production pace, and therefore wastewater treatment needs, was variable, and there was inconsistent record-keeping at the pre-treatment plant, normalization using recorded flow logs was not possible. Operator-set effluent discharge rates, as observed by researchers, varied widely from a low-discharge rate of around 20,000 gallons per day during periods of low cheese production and/or drought, to a high-discharge rate of 50,000 – 60,000 gallons per day during periods of high cheese production and/or precipitation. Accordingly, the filter experienced periods of high, medium, and low flow conditions, with a consequent wide variation in hydraulic residence times, though these exact times are unknown. Researchers were told that at times filter influent was stopped completely for up to 5-6 days at a time, with only 1-2 days of flow, though this information is not verifiable with flow records. It is mentioned to illustrate the variability experienced during operation. Therefore, a limitation of this study was that control of the residence time for wastewater within the filter was not consistent across the study period. Despite the variability with respect to flow regime and hydraulic residence time, the filter was still able to achieve a substantial reduction in TP. With greater control of influent rate, and therefore hydraulic residence time, it is possible that these rates could be further optimized, especially with respect to controlling increases in pH. However, the variable flow conditions present at the study site may be relatively typical of sites where a similar filter system might be considered for implementation.

#### **2.5.12 Feasibility of Scaling Filter to Accommodate Full Wastewater Flow**

In order for the cheese production facility to realize the full benefit of an EAF steel slag

filter for P treatment for all of its wastewater, the filter size would have to increase to accommodate, at minimum, the average minimum discharge volume of 20,000 gallons per day ( $75.7 \text{ m}^3/\text{day}$ ). Given that EAF steel slag has a pore volume of ~40%, a total filter volume of 34,500 gallons ( $130.6 \text{ m}^3/\text{day}$ ) would be necessary to fully treat all discharged wastewater with a hydraulic residence time of 24 hours. A filter of this volume at a depth of ~5 feet (1.5 m) would require a land area of  $970 \text{ ft}^2$  ( $90 \text{ m}^2$ ). EAF steel slag has been demonstrated to have a P saturation capacity of 3.65-5.00 g P / kg slag (Drizo, 2008). Steel slag has an average density of 2.25 tons /  $\text{m}^3$  (Euroslag Association, 2016). Therefore the system would have ~300 tons of EAF steel slag with a P saturation potential of ~1500 kg total. Assuming  $75.7 \text{ m}^3/\text{day}$  effluent at 24 mg/L P concentration, the daily P load would be 1.8 kg P / day or 657 kg/P annually. The material lifespan would therefore be between ~800 days or just over 2 years before replacement would need to occur. However, according to Drizo and Picard (2014) when treating larger volumes of wastewater (greater than  $10 \text{ m}^3/\text{day}$ ) the filter should consist of a replaceable filter unit, main filter unit, and a pH adjustment unit. If instead of a single  $90 \text{ m}^3$  volume filter the configuration consisted of a  $25 \text{ m}^3$  replaceable filter unit (RFU), followed by  $75 \text{ m}^3$  main filter unit, then material could be exchanged from a RFU after reaching saturation, prolonging the life span of the main filter unit. Scaling up to this size at this facility would be possible given the available space and the replacement period is reasonable.

## **2.6 CONCLUSIONS**

EAF steel slag has the potential to reduce TP concentrations in cheese production facility wastewater by ~90% even if hydraulic residence time is highly variable. EAF steel slag

similarly shows promise for reducing DRP concentrations. In future applications of such a filter, it would be highly recommended that influent rate, and therefore hydraulic resident time of wastewater to EAF steel slag, be more tightly controlled to produce optimal results with respect to P removal versus pH.

The only filter compartment in this study to have a significant beneficial effect on either TP or DRP concentration was Compartment 2, containing EAF steel slag. Compartment 1 had no significant effect on TP or DRP. Compartment 3 actually caused a slight increase in DRP, but had no significant effect on TP. Only Compartment 2 had a significant effect on TSS. This indicates that EAF steel slag is effective at solids filtration, as well as P removal.

Though Compartment 3 did have a significant beneficial effect on pH, reducing it nearly one order of magnitude, the filter's effluent pH average was still higher than admissible water quality standards in Vermont. Elevated pH was thought to be largely due to filter stagnation. Future applications of this filter media should ensure that either continuous design flows are maintained or alternate pH adjustment techniques are used to ensure that effluent meets applicable water quality standards.

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### **3 METHODOLOGICAL CONSIDERATIONS FOR SAMPLING OF UNLINED BIORETENTION SYSTEMS IN A FIELD SETTING**

#### **3.1 KEYWORDS**

Green stormwater infrastructure, monitoring infiltration, water quality, bioretention

#### **3.2 ABSTRACT**

Two unlined infiltration bioretention practices in Vermont were monitored for total suspended sediment (TSS), total phosphorus (TP), and dissolved reactive phosphorus (DRP) in influent and effluent. Pre-BMP construction samples were obtained for one site in Waitsfield, VT from an existing catch basin rim to characterize runoff at that site. Samples of effluent were taken from perforated underdrain pipes located beneath the soil media either by grab sampling using a hand pump or by automated sampling. No effluent samples were obtained using hand-pump-assisted grab sampling at the bioretention practice in Waitsfield, VT, while some samples were obtained using automated sampling of the underdrain from the bioretention practice in Hardwick, VT. This paper discusses the sampling methods used at each of these sites and how they could potentially be improved to obtain more and higher quality samples through a storm event to better characterize water quality in systems that are not necessarily designed to be monitored. This discussion will attempt to provide guidance for practitioners looking for methodologies suitable to sampling from green stormwater infrastructure practices that use infiltration as the primary method of treatment.

#### **3.3 INTRODUCTION**

Vermont is facing numerous water quality challenges associated with runoff and development, principal among them the need to reduce phosphorus (P) loading from land

surfaces to waters of the State, the need to reduce or eliminate channel-altering flows from developed surfaces, and the need to mitigate flooding associated with larger storm events. The State's largest body of water, Lake Champlain, is under a Total Maximum Daily Load (TMDL) agreement with the US EPA, administered by the VT Department of Environmental Conservation (VT DEC) to reduce P loading to the Lake . The VT DEC has embraced the use of green stormwater infrastructure to help accomplish this (VT ANR, 2016). However, assuring that reductions in P loading are occurring is a critical part of the TMDL agreement. Typically, reductions are assumed using modeling procedures that rely on published data for pollutant washoff rates, as well as typical removal rates for different GSI practices. Very little site- and practice-specific monitoring occurs.

During the summers of 2012 and 2013 two bioretention practices in Vermont were monitored for water quality. Our objective was to develop a relatively simple method by which managers and regulators could monitor performance of infiltration-based GSI practices that were not originally designed with monitoring infrastructure built-in. Such monitoring infrastructure can be quite expensive and includes equipment like automated samplers that use flow-based or time-based sampling to assess bioretention performance as well as sumps, inlet and outlet weirs, and other hard infrastructure that is aimed at monitoring and not included in most bioretention design recommendations. More successful monitoring programs also typically employ a liner surrounding the bioretention system in order to capture effluent water for comparing water quality with influent; this clearly countermands the purpose of the bioretention practices that are intended to be unlined and infiltrate runoff (Ballesterio, et al., 2012; Dietz & Clausen, 2005; Hsieh &

Davis, 2005; Lucke & Nichols, 2015; Mangangka, et al., 2015). Neither of the practices in this study were explicitly designed to be monitored. The bioretention in Waitsfield had three separate curb cut inlets fitted with special inlet-protection structures for sampling runoff entering the BMP. Additionally, sampling prior to the construction of the bioretention was undertaken to characterize the parking lot's runoff. These samples were obtained from the lip of an existing catch basin. For effluent sampling, the bioretention was designed with an underdrain and cleanout that was used for sampling. The bioretention in Hardwick had perforated pipes with sampling ports designed and installed at the end of the project design process. This sampling infrastructure was never part of the original practice and so represents an ad hoc method. We undertook these studies to attempt to collect and analyze influent samples for total suspended sediment (TSS), total phosphorus (TP), and dissolved reactive phosphorus (DRP) from curb inlets (Waitsfield) or inlet pipes (Hardwick) and effluent samples from the perforated pipes, or underdrains, located under the bioretention soil media. In Waitsfield, the underdrain was enclosed in a 1.5" (3.8 cm) washed crushed stone jacket. Underdrains at Hardwick were placed directly in soil media. The underdrains were accessed via a vertical a Tee-junction pipe fitting that emerged above the bioretention soil surface to function as a sampling port.

This article will talk about each site's design, as well as the challenges associated with sampling both influent and effluent at the respective sites. Both practices were unlined bioretention systems, meaning that any runoff entering the bioretention ponding area was designed to infiltrate through soil into groundwater. In Waitsfield, a perforated pipe underdrain with a cleanout was included to prevent nuisance ponding during periods of

heavy precipitation and soil saturation. Underdrains of this type are usually only included for this reason, or in cases where high groundwater may occur. In Hardwick, perforated pipe underdrains were included in an attempt to facilitate side-by-side testing of two different soil media types. The practice was never originally intended to have perforated pipe underdrains installed.

The first bioretention project Village Square shopping center in Waitsfield, VT, was monitored during the summer of 2012; influent was captured, but no effluent sample was obtained from the bioretention's perforated pipe sampling port for any storms, which perhaps relates to the infiltration rate into the native soil, or to sampling protocol challenges. Thus no comparison can be made with respect to water quality between influent and effluent, though some characterization of influent water quality was possible.

The second bioretention project, located in a residential street right-of-way in Hardwick, VT, was monitored during the summer of 2013. Both influent and effluent samples were obtained, but uncertainty exists as to whether or not meaningful comparisons between samples can be made, due in part to equipment malfunctions and sampling errors, as well as implicit effects of infiltration.

The challenge associated with monitoring these projects are likely to be faced by many municipal storm sewer system administrators and regulators with respect to practices that are designed to treat runoff via infiltration, but not designed specifically to monitor water quality. Questions emerge, including: how does one obtain a sample of the 'treated' water? At what point during a storm event is that sample obtained? Do the samples fairly represent

changes in water quality in a system that can be attributed to infiltration-based bioretention performance?

### **3.4 METHODS**

#### **3.4.1 Village Square bioretention – Waitsfield, VT - Description of Research Site**

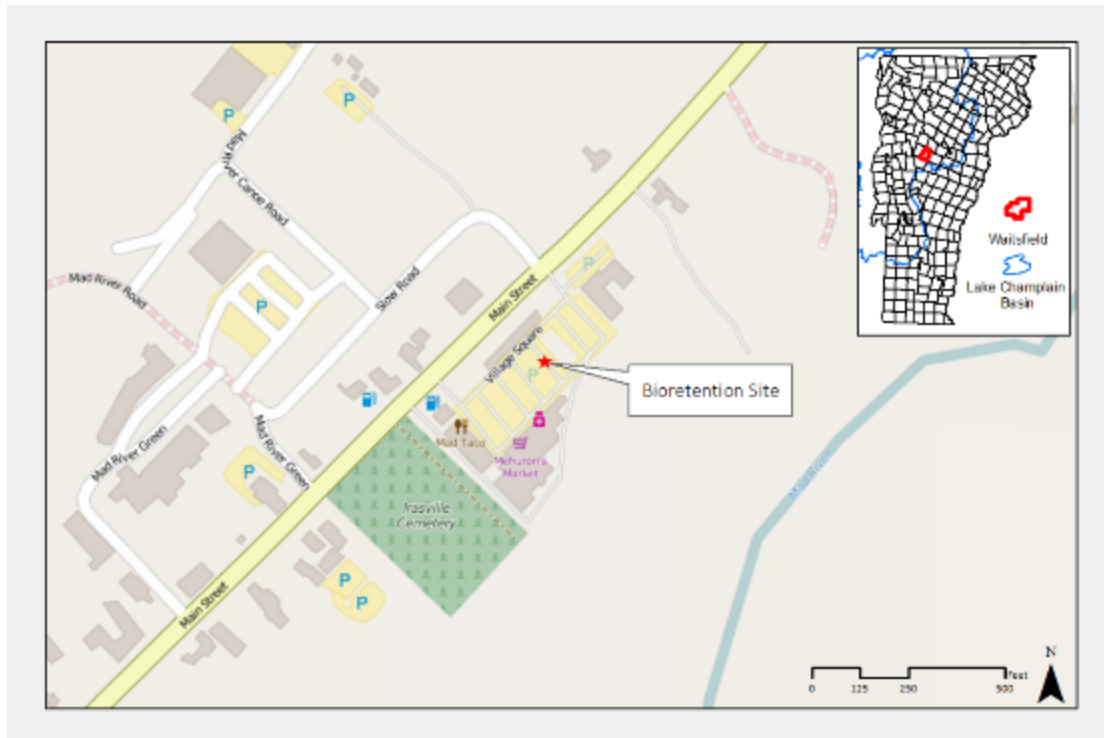


Figure 3-1: The Village Square bioretention site in Waitsfield, VT is located in the Mad River watershed within the Lake Champlain Basin.

The Village Square bioretention sampling case study site is located in a commercial parking lot in Waitsfield, VT (Figure 3-1). Waitsfield is part of the Mad River Valley, a tributary of the Winooski River in the Lake Champlain Basin. The drainage area for the Village Square bioretention is 1.008 acres as delineated using 1-foot contours generated from site-specific topographic survey using a total station survey instrument (Figure 3-2).



95.34% of this area is classified as impervious (pavement, concrete sidewalks, asphalt roads, and rooftops). Only 0.047 acres is pervious, consisting of a small patch of mowed grass. The parking lot serves a grocery store, as well as several small businesses and restaurants.

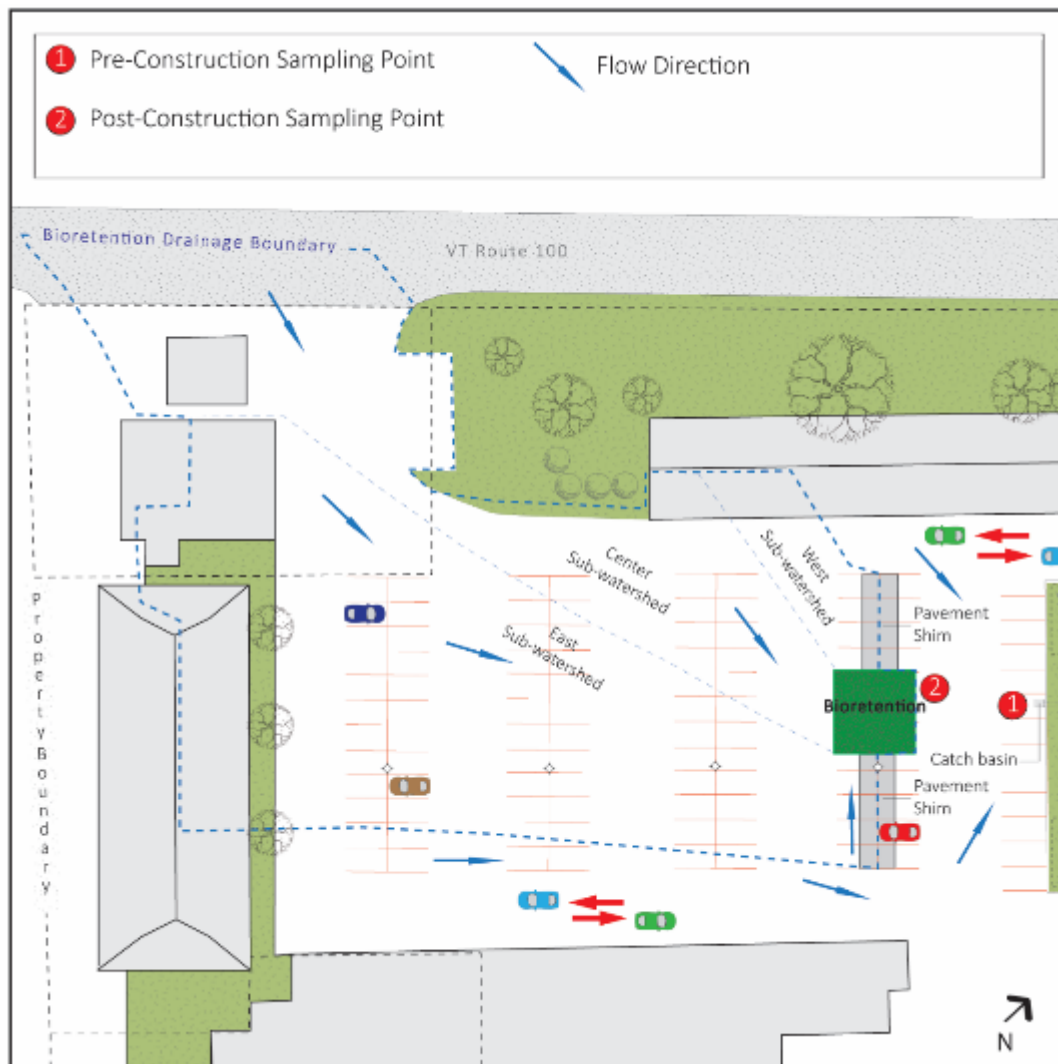


Figure 3-2: Village Square bioretention site plan showing drainage and traffic patterns, as well as sampling point locations.

The bioretention practice is 900 ft<sup>2</sup> and occupies the space of approximately eight parking spaces. There are three inlets to the bioretention on the east, west, and south-east sides. These inlets are protected from inlet scour using RainGuardian™ (Anoka Conservation District, 2011) inlet devices that feature a steel grate at grade to filter out large floatables (trash, leaves, etc.). The grate drops into a small sump with a screened inlet out into the bioretention – this feature serves to both allow for some solids settling and screening. The parking lot, as originally configured, allowed nearly all runoff to collect via sheet flow to a single catch basin inlet at the northern edge of the lot (Figure 3-2). To ensure that runoff would enter the bioretention, shallow pavement shims (analogous to very flat speed bumps, maximum 3” (7.62 cm) tall, and 18’ (5.4 m) wide) were retrofit into the parking lot surface to re-direct runoff into the east and west inlets (Figure 3-2). These shims are shallow enough that cars can drive over them without incident. They are also configured so as not to interfere with winter plowing activities.

Calculations using HydroCAD v.10.0 build 12 shows the total runoff volume for the site to be 2,875 cubic feet (21,506 gallons) for the 0.9” / 24-hour water-quality volume (WQv) storm and 5,575 cubic feet (47,708 gallons) for the 2.04” / 24-hour channel protection volume (CPv) storm. The bioretention is sized to completely contain and infiltrate the CPv over the course of 48 hours. Runoff events that exceed this volume are bypassed via a 24” circular grate overflow that outlets to the existing storm sewer system.

#### **3.4.2 Sampling Pre-Construction**

Nine pre-construction samples were collected using grab samples between April 23, 2012 to September 18, 2012. Samples were analyzed for TSS, TP, and DRP. Efforts were made

to collect grab samples at different times during each precipitation event – beginning, middle, and end. Samples were collected at the existing catch basin rim using an acid-washed 250-ml Nalgene bottle that was triple-rinsed using reverse-osmosis distilled (RO) water and runoff water prior to collecting the sample. The contents of this bottle were then transferred to a 1-L Nalgene bottle that was acid-washed and RO water triple-rinsed to obtain adequate sample volume for P and TSS testing. This bottle was then placed in an insulated cooler with ice and held at 4° C for the 45-minute transportation to the University of Vermont’s Agriculture and Environmental Testing laboratory.

Pre-construction sampling to characterize runoff pollutants could be an important tool for managers, regulators, and designers in the future as governance moves towards specific concentration discharge limits for stormwater management. Clark & Pitt (2012) note that sampling a site to characterize specific pollutants will better help designers manage them with specific solutions.

### **3.4.3** *Post-Construction Sampling*

Following bioretention construction, two separate sampling locations were intended to be utilized, one for influent, the other for effluent. The influent samples were to be obtained from each RainGuardian using the same grab sampling methods described in the Pre-Construction Sampling section. Samples were to be obtained from the lip of the RainGuardian by removing the horizontal metal inlet grate above the RainGuardian’s initial sediment sump. Removing this grate creates a drop of approximately 4” (10.16 cm) from the lip to the invert of the sump which facilitates grab sampling. This procedure was

to be repeated for each of the three RainGuardian inlets and the samples composited for analysis.

The effluent samples were to be obtained from a 6" (15.24 cm) perforated filtrate sampling pipe located 4' (1.22 m) underneath the surface of the bioretention (Figure 3-3). This perforated pipe has a 6" (15.24 cm) vertical standpipe connected to the horizontal perforated pipe via a Tee-fitting and is capped approximately 6" (15.24 cm) above the bioretention's surface. The cap can be removed to allow for the insertion of a 0.25" (0.64 cm) suction hose attached to a hand-operated vacuum pump to bring water up from the sump of the pipe into a 1-L Nalgene bottle prepared as described in the Pre-Construction Sampling section. Additionally, the pump hose and pump internals were acid-washed and tripled-rinsed using RO water prior to each sampling event. If water was present in the filtrate sampling pipe, 250 ml of sample would be run through the hose and pump to rinse it prior to collection the 1-L Nalgene. All samples were stored, transported, and analyzed as described in the Pre-Construction sampling procedure above.

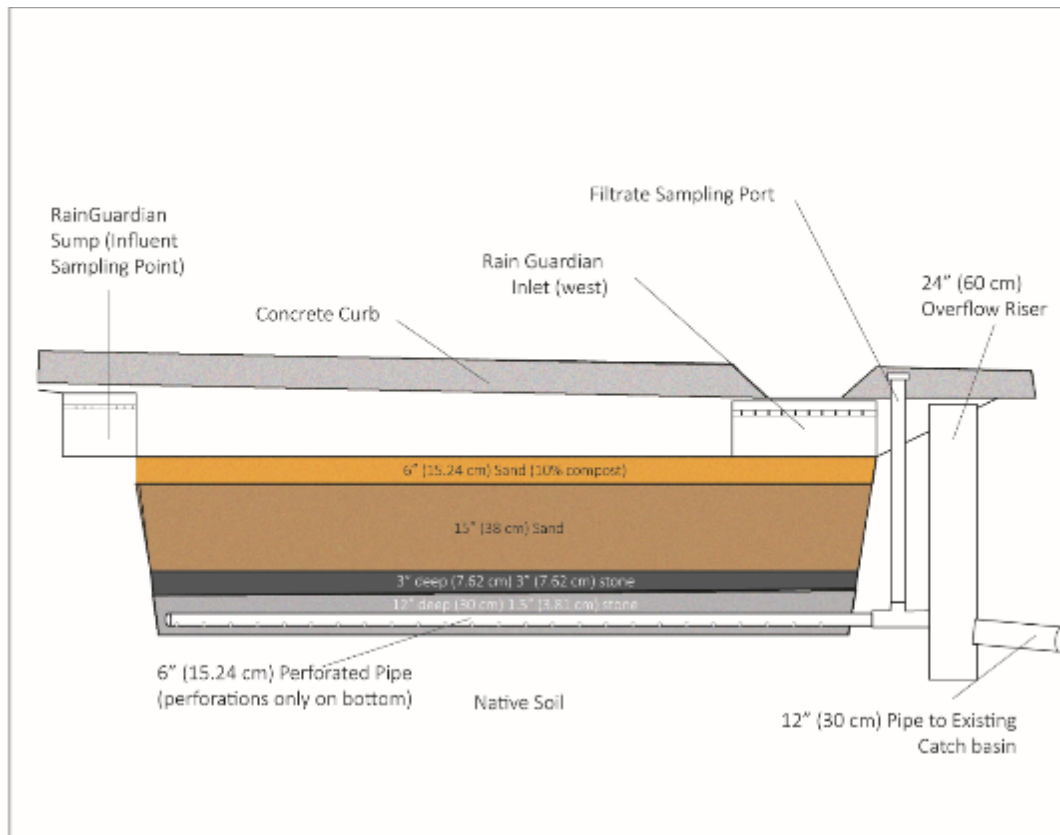


Figure 3-3: Section view and sampling infrastructure for the Waitsfield Village Square bioretention facing west.

### 3.5 Cherry Street bioretention – Hardwick, VT

#### 3.5.1 Description of Research Site

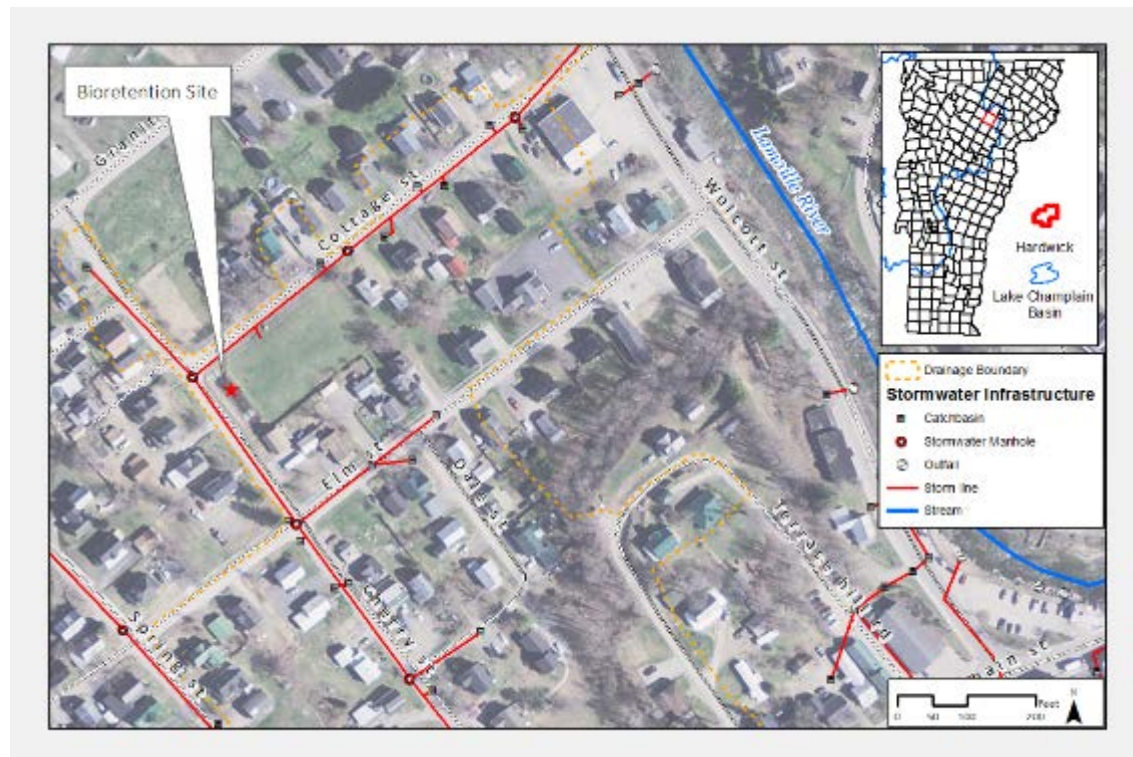


Figure 3-4: Hardwick bioretention site on the corner of Cherry and Cottage streets. The bioretention site is located on a piece of publically owned land. Stormwater infrastructure is shown – a short lateral from the main (red) stormwater line along Cherry Street connects to a flow-splitter that inlets to the bioretention.

The town of Hardwick, VT, is a small village located in northeast Vermont. It has a small, urbanized center with light commercial development, surrounded by residential neighborhoods. The Cherry Street bioretention project is located on town-owned land adjacent to a community playing field (Figure 3-4). The bioretention project receives water from a closed pipe system that outlets to the Lamoille River, a tributary of Lake Champlain, which runs through the center of town. It was retrofitted into the site following a study conducted by the VT Department of Environmental Conservation on potential stormwater

BMP locations. A flow-splitter was installed to divert an 18,400 ft<sup>3</sup> (~137,600 gallons) WQv from the 24.5-acre watershed, 7.8 acres (31%) of which is classified impervious.

### System Design

The Cherry Street bioretention in Hardwick was designed as a paired bioretention system, sharing a combined influent pipe and forebay, but splitting the flow into two linear bioretention cells using two 6" (15.24 cm) HDPE pipes with a higher stone spillway (Figure 3-5).

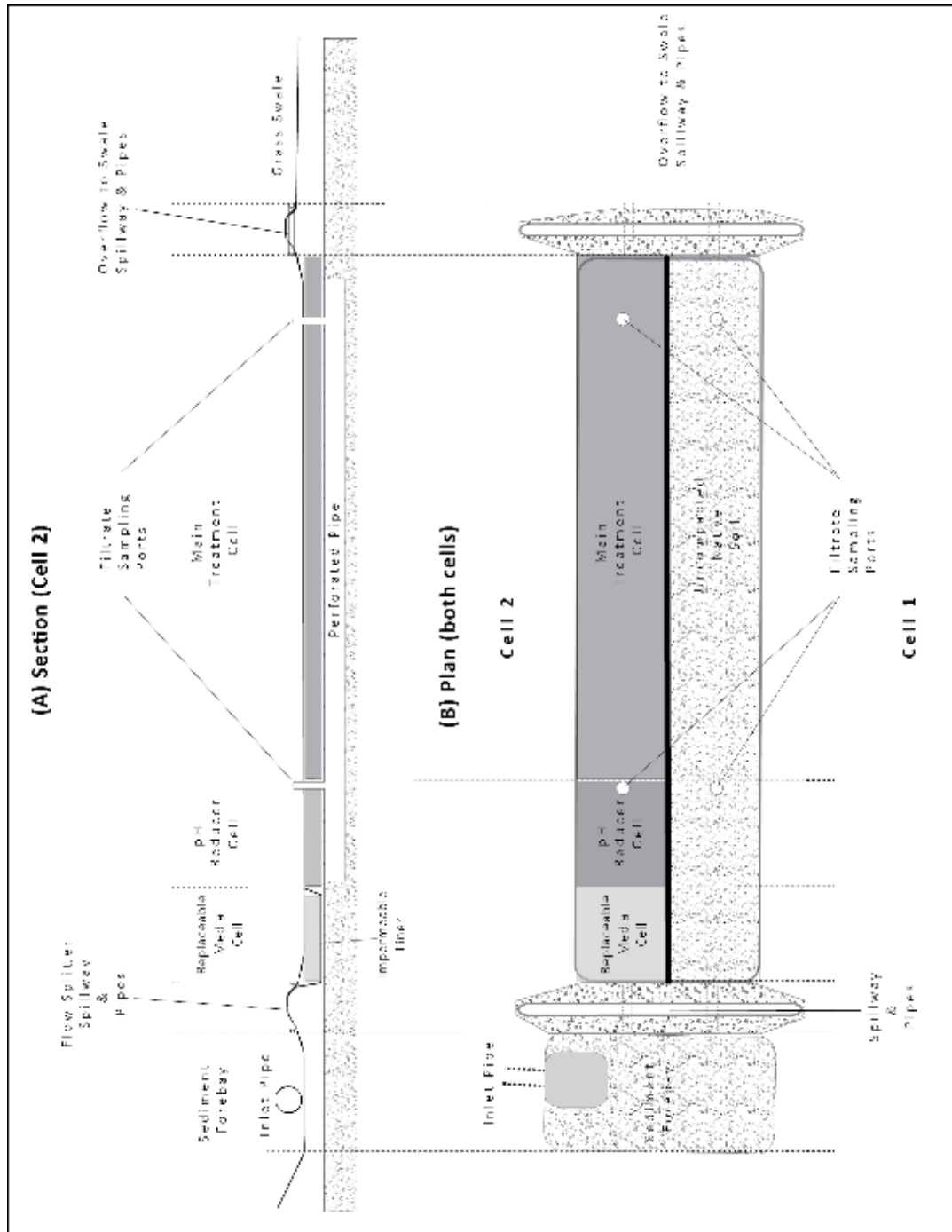


Figure 3-5: A) Section and B) Plan views of the paired bioretention cells at Hardwick. A) shows the section for Cell 2 and its sub-cells only. Cell 1 is composed only of native soil and 6" (15.24 cm) of topsoil. Samples were only taken at the second set of filtrate sampling ports (nearer the overflow to the swale).



The sediment forebay is approximately 15'x20' (4.57x6.09 m) wide. Each cell is approximately 10' (3.04 m) wide at the ground surface and 80' (24.38 m) long and will pond up to 12" (30.48 cm) before reaching a 6" (15.24cm) HDPE pipe that serves to split the flow. An additional stone spillway is located 9" (22.86 cm) above the HDPE pipe as a level spreader. The treatment cells both outlet to a low-gradient grass swale that further treats runoff before directing any excess back into the closed drainage system via a 12" (30.48 cm) HDPE pipe. A vertical impermeable plastic barrier separates the two sides of the bioretention to a depth of 3' (0.91 m) to prevent infiltrated runoff from co-mingling. Over the course of 6 storms over 0.5" (1.27 cm) and one of 1.4" (3.56 cm), as well as nearly 40 storms at or under 0.5" (1.27 cm) storms from May 1 to August 28, 2013, the pipes, stone spillway and grass swale were never observed to have overflow water in them, indicating that the system was adequately sized for the diverted WQv and that diverted water was completely infiltrating.

The composition of soil media used to filter runoff differed between the two cells. Cell 1 uses only native soil (Adams-Nicholville urban land complex) and 6" (15.24 cm) of topsoil with an average 20% organic content; there are no additional soil amendments. The cell is completely unlined. Cell 2 is configured with three sub-cells in longitudinal sequence (Figure 3-6).

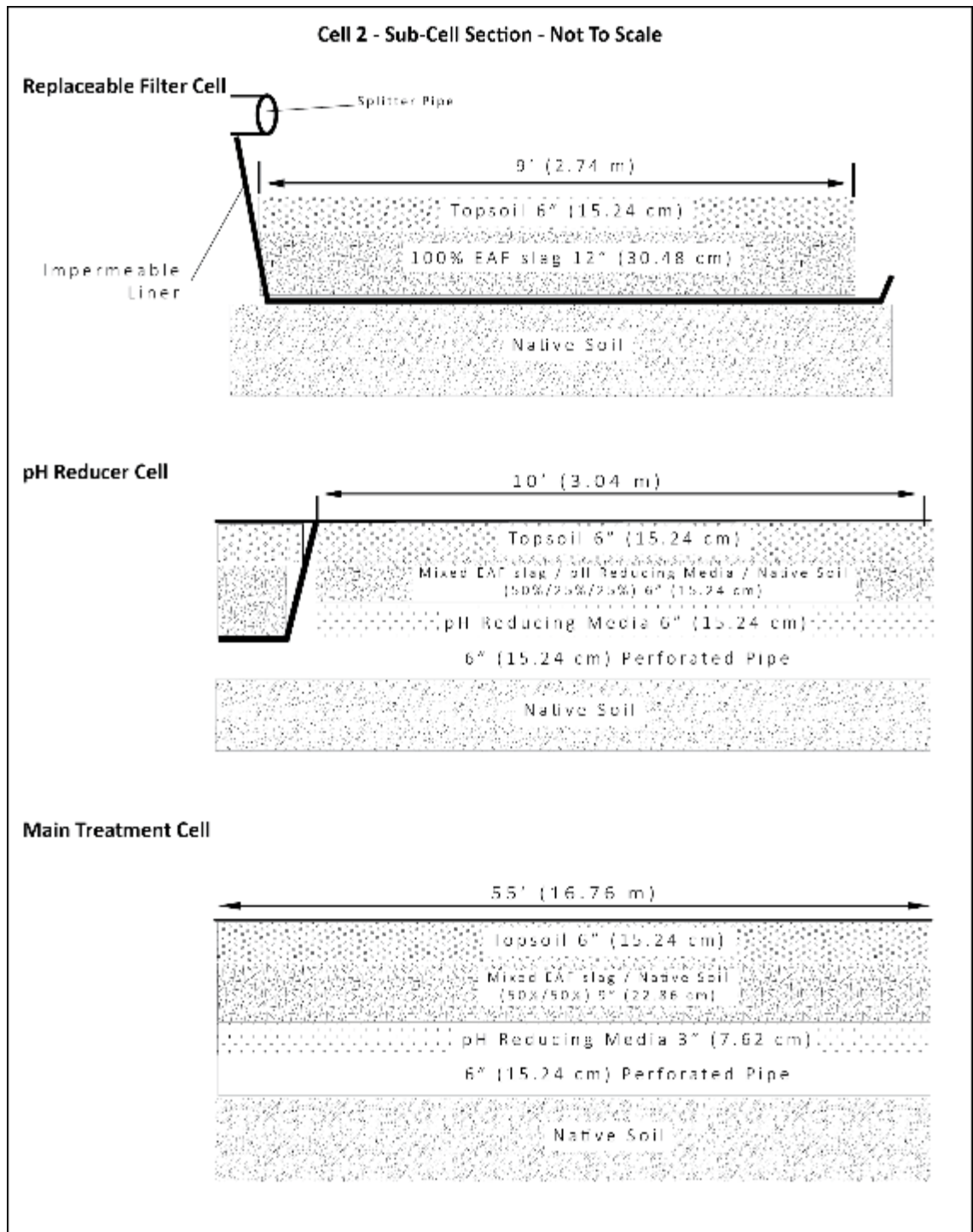


Figure 3-6: Cell 2 showing section views for each sub-cell, beginning with the replaceable filter cell, which is lined, to the subsequent pH reducer cell and finally the main treatment cell.

The first sub-cell, called the replaceable filter cell, is 9' (2.74 m) in length and composed of EAF steel slag in poly-mesh bags for easy installation and removal. These bags are stacked to a depth of 12" (30.48 cm) and underlain with an impermeable plastic liner to encourage horizontal flow-through. 6" (15.24 cm) of topsoil covers the entire sub-cell. The second sub-cell, the pH reducer cell, is 10' (3.04 m) in length and was designed to reduce the pH of runoff filtered through the replaceable filter cell and consists of a 50/25/25% (by volume) mix of EAF steel slag, native soils and cedar wood chips, covered with 6" (15.24 cm) of topsoil. The pH reducer cell follows the replaceable filter cell as the potential contact time between runoff and EAF steel slag in this cell could potential elevate pH levels in filtered water. The main filter cell section is 55' (16.76 m) in length and includes 9" (22.86 cm) of 50/50% (by volume) mix of EAF steel slag and native soil, underlain by final 3" (7.62 cm) layer of cedar wood chips, and covered with 6" (15.24 cm) of topsoil.

### **3.5.2** *Sampling*

Sampling was accomplished using Teledyne ISCO 6712 autosamplers. Influent samples were collected from the influent pipe (12" (30.48 cm) HDPE) while effluent samples were collected from the 6" (15.24 cm) HPDE perforated sampling pipes located 3' (0.91 m) under the soil surface of each of the bioretention cells (Figures 5 and 6). Water infiltrated into the soil, entered the pipes via 0.25" (0.64 cm) perforations and collected in the bottom of the pipes. Each filtrate sampling pipe was capped fully at the influent end and partially capped at 3" (7.62 cm)) on the effluent end to accomplish necessary 'ponding' of water in the pipe for sampling (Figure 3-7).

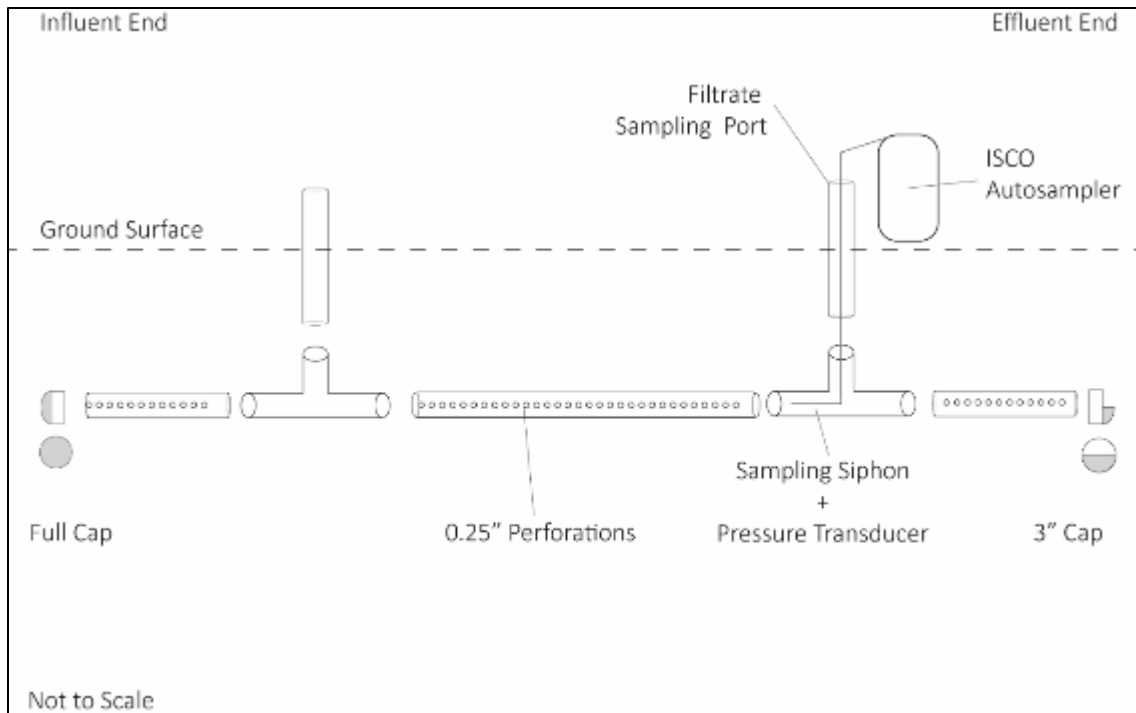


Figure 3-7: Sampling port configuration showing the capped influent end and half-capped effluent end of the pipe, with vertical, non-perforated sampling ports where the ISCO sampling tubes and pressure sensors were inserted. Vertical pipes were also capped except for small holes allowing for passage of sampling equipment. The extent of the perforated pipe and sampling port locations are shown on Figure 5. Note that samples were not obtained from the first sampling port – only from the second.

Each of the three ISCOs was programmed to sample full, discrete 1-liter samples via flow-weighted sampling during storm events. However, due to the wide variability of precipitation volumes, and the relative unpredictability of runoff volumes from each storm, time-enabled versus flow-weighted sampling was eventually chosen as the most reliable method. Typically, samples were obtained on 1-hour intervals if storm durations were predicted to be less than 24 hours. For storms predicted to last more than 24 hours, increments of up to 2 hours were used.

All 1-L plastic bottles in the ISCO unit were acid-washed and tripled-rinsed with RO water prior to placing them in the ISCO for sample collection. Unlike with grab sampling, bottles were not triple-rinsed with sample prior to collecting final sample as this was not possible with the ISCO autosampler.

### **3.5.3** *Sample Collection Events*

During the spring and summer of 2013, there were 38 precipitation events above 0.1". Events over 0.1" of precipitation were required for sampling as events below this threshold did not fill and overflow the commingled forebay – no runoff water entered either cell, so no sample was possible via the perforated pipes and ISCO sampling equipment. Of these 38 events, five discrete events were sampled in some way over the course of twelve days of precipitation. These events were 5-23, 5-29 – 5-31, 6-11 – 6-13, 7-8 – 7-10, and 8-3. These events represent 3.06" of precipitation cumulatively out of the total 15.68" that fell during the period from 5-1 to 8-31, or 13% of total possible sampling events.

Of these five discrete events, influent and effluent samples were only possible in two of them: 5-29 – 5-31, and 7-8 – 7-10. The other three events had only either influent or effluent sampled. This was either due to sediment clogging of the influent intake probe, sample-enabling failure of the ISCO probes, or sample-bottle misalignment in the ISCO bottle carousel. These equipment issues were compounded by the distance from the laboratory to the field site – at approximately 1.5 hours one-way, ensuring that field personnel were on-site for the start of a precipitation event to ensure that equipment was functioning properly was problematic.

Samples were collected at the single influent ISCO, as well as from the two effluent ISCOs stationed at the second set of filtrate sampling ports. The remoteness of the site from the lab facility made storage of samples at 40 C at all times impossible and the length of certain sampling periods precluded the maintenance of sub-24 hour hold-times for all samples.

### **3.6 Discussion**

#### **3.6.1 Influent Sampling**

Sampling at the Village Square site of pre-construction storm events did provide some data related to TSS, TP, and DRP concentrations in the parking lot's runoff, but this data is of limited use. While concentrations can be analyzed from the grab samples obtained, these concentrations are snapshots only, providing a glimpse of a very finite portion of a storm's runoff total pollutant concentration range. They do little to characterize the overall range of pollutant concentrations possibly found during the entire storm. Also, the lack of verifiable flow data precludes any development of mass loading estimates. While using the approximate watershed size and land use type could in theory lead to the development of a volume discharge, this method is not preferred to actual measurement of flow volumes.

The limitations associated with using only grab sampling at this site suggest that this method should not be employed; alternative methods for retrofitting sample and analysis equipment into an existing site in order to gauge runoff quality or quantity prior to implementation of Green Stormwater Infrastructure (GSI) projects would likely result in more reliable data and straightforward data collection methods. In the case of the Village Square parking lot, it may have been possible to obtain better pre-construction runoff samples by retrofitting a weir into the existing catch basin sump and using a level sensor

and auto-sampler to obtain flow-weighted samples throughout individual storms. However, there are limitations associated with this method, namely the need to monitor this equipment closely during operation. As the distance between the research site and the laboratory was 45 minutes by car, this type of monitoring would have been difficult. Perhaps a more effective means of pre-construction monitoring would have been to employ a sensor to measure for variables such as turbidity and pressure in the sump of the catch basin. Some researchers have used turbidity as a proxy measurement for phosphorus, though even this method still relies on obtaining actual samples in order to develop the proper mathematical relationship between measured turbidity and analyzed phosphorus (Christensen, et al., n.d.; Galloway, 2006; Kim, et al., 2012; Yamamoto & Suetsugi, 2006).

Samples of influent taken at the Hardwick site were obtained using an ISCO auto-sampler from the shared influent pipe. Samples were obtained using time-based sampling, which did result in obtaining samples from numerous storms. However, it was difficult to calculate flows as the influent pipe was not fitted with a weir, which would have made gauging discharge volumes more accurate. Additionally, it was frequently found that sediment washed off impervious surfaces would completely cover the intake tube and pressure transducer, skewing the pressure reading and rendering sampling impossible. This should be taken into account for future influent sampling endeavors, especially in larger drainage areas where large volumes of sediment are possible.

The largest issue with influent sampling at the Hardwick site was that all influent samples were taken at the outlet of the storm sewer pipe that fed the forebay. Runoff then entered the forebay, where some of the volume would infiltrate and sediment would settle out, and

only when runoff ponded up to the level of the 6" (15.24 cm) HDPE splitter pipes, or over the stone spillway, would it actually split and enter the bioretention cells. Therefore any pre- and post-treatment sample comparison ignores the effect that the forebay may have had on pollutant removal. A better option would have been to install auto-samplers on each of the overflow pipes leading to the bioretention cells. This was not done due to equipment and budget constraints. Even if this had been done, there still would have been inaccuracy associated with sampling at the overflow pipes as runoff was also free to pass through the pore spaces in the stone spillway separating the forebay from the two bioretention cells.

It is clear that, while some success was had in sampling influent at Hardwick, modifications to the treatment design would have to be made to obtain a more representative sample of runoff entering each bioretention cell. First, installing a weir at the inlet pipe to more accurately gauge influent would have been more effective. Secondly, it would have been ideal to install ISCO samples on each of the splitter pipes to more accurately gauge flow to each Cell, as well as measure pollutant concentrations for each. Ensuring that each stone spillway was impermeable would be critical for this method. Installing a soil moisture sensor in each Cell at the depth of the perforated pipe would be beneficial to this sampling design, as that time-stamped information would allow researchers to know when runoff had infiltrated to the depth of the pipe, thereby establishing an approximate time between runoff entering the cells and runoff infiltrating to the sample depth. This could have led to the development of system for starting sampling with the effluent ISCOs. Finally, adding additional lateral pipes from the central perforated pipes, to obtain more extensive coverage of the bioretention Cells to capture more infiltrated runoff may have led to more success.



### **3.6.2** *Sampling Effluent from an Infiltration Practice*

No samples were obtained from the site in Waitsfield, despite five different attempts to pump water from the perforated pipe underneath the bioretention practice. This is possible due to three factors: (1) water was preferentially infiltrating into the well-drained native soils surrounding the bioretention system and therefore not present in the underdrain, (2) timing with respect to storm events, and (3) the perforated pipe placement, extent, and outlet design.

If the first factor explains the lack of ability to collect samples, it may also be an indicator of the design's success at infiltrating stormwater; unfortunately, because the other two factors are also plausible, there is no way to know for certain that infiltration to surrounding soils was the primary explanation for the underdrain pipe being empty.

With respect to timing, the issue is one of logistics. In order to obtain an influent sample using simple grab-sampling methods, it is generally advisable to be on-site to obtain that sample as runoff starts to enter the practice and pond up (Cha, et al., 2016; Lee, et al., 2002; Sansalone & Cristina, 2004). This will typically yield a sample of the first flush and give some indication of the practice's total effectiveness at removing pollutants. However, this recommended timing is counterintuitive if the goal is to obtain an effluent sample from the perforated pipe as, due to engineered retention times, runoff has not yet had a chance to infiltrate to the level of the pipe at the start of a storm event. Another way to proceed would be to bury a soil moisture sensor at the depth of the perforated pipe which could have indicated the moment that runoff had infiltrated to the depth of the pipe (Kwiatkowski, et

al., 2007). This would have at least allowed researchers to know if water should have been entering the pipe, rather than simply waiting.

The second issue is one of design. At the Waitsfield site, the perforated pipe outlet into the 24" riser structure could have been fitted with a weir or dam to cause ponding in the pipe for monitoring purposes. A weir or dam may have resulted in adequate water within the pipe to sample using a siphon tube. This same method was employed at the site in Hardwick with limited success however, so using such a technique is recommended with reservation.

At Hardwick, a single 6"-perforated pipe with sampling ports daylighting above the soil surface underlay each of the paired bioretention cells. These pipes were fully capped on one end, and capped to 3" on the 'outlet' end, which did result in ponding in the pipe. However, attempting to sample using flow-weighted analysis was not possible given the lack of a weir, as well as the lack of horizontal flow within the pipe. The only possible measure at Hardwick was pressure above the transducer and sampling intake tube. However, this was inadequate to sample the beginning, middle, and tail of a storm event. Using time-based sampling at this site was eventually determined to be the best alternative, but without an accurate measurement of flow, it is difficult to determine where each of these samples might fall along an individual storm's hydrograph/pollutograph. A potentially better option given conditions at this site would have been to obtain composite samples from infiltrated runoff, resulting in an average concentration across the storm duration (Hatt, et al., 2009; Kim, et al., 2005; Law, et al., 2008).

### **3.6.3** *Extensive Perforated Pipes for Sampling*

Another option, would be to use a more extensive perforated pipe system under the bioretention's soil media. Hussain, et al., (2016) used this method to monitor a 3-acre 'dry pond' in Minnesota. The system of pipes underneath the dry pond was extensive, consisting of 140' of 8" perforated trunk lines with 480' of 4" perforated laterals arranged in symmetrical V-formation on either side of the trunk lines. All trunk lines joined together at a common outlet. A weir and autosampler at the inflow, coupled with an autosampler and pressure transducer at the outlet, provided estimated of volume in and out. Some volume was lost to infiltration, while the rest flowed out the outlet. The researchers were able to obtain samples to analyze in this way. However, this method for sampling, while resulting in samples for analysis and some characterization of performance, is misleading. Because the perforated pipes exist and drain to a common outlet, runoff that would otherwise have infiltrated through additional soil layers potentially receiving further treatment, is captured by the pipe network. Peak discharge reduction is decreased, pollutant removal benefits are reduced, and groundwater recharge is lessened. While effective for monitoring, this configuration may not be the most effective for stormwater treatment.

### **3.6.4** *Groundwater Monitoring Wells for Sampling*

Rather than use perforated pipes in any configuration for sampling of infiltrated runoff, another possibility at this site would have been to use a monitoring well. Monitoring wells are used for obtaining samples of groundwater to determine pollution levels. However, there have been some instances where monitoring wells have been used to monitor infiltration runoff. In 1986, Ku and Simmons were among the first to publish results of a study from Long Island that monitored the effectiveness of 'recharge basins' (essentially

infiltration basins treating stormwater from parking lots) at pollutant removal using monitoring wells (Ku, et al., 1984). Ku, et al., (1984) determined typical ground water levels prior to initiating the study, then used a sensor to indicate when infiltrated water reached that depth. Samples were obtained from the “water mound” produced by the infiltrated stormwater. Seasonal and storm-influenced variations in groundwater should also be accounted for in such a method; monitoring of groundwater pre and post implementation a water treatment system should make every effort to control for these types of variability.

Despite its potential, it does not appear that this method has been widely adopted. The Urban Stormwater BMP Database in its guidance on monitoring BMPs, references using monitoring wells, and even recommends a method developed by Granato & Smith (1999) using what those researchers termed a ‘Robowell,’ or automated monitoring well sampling system, but does not provide any additional specific guidance.

Nielsen, in the 2007 book *The Essential Handbook of Ground-Water Sampling*, notes that obtaining truly ‘representative’ samples using monitoring wells can be complicated (Nielsen, 2007). Samples can be altered due to two common practices in ground water sampling: well purging and well screening. Prior to obtaining a sample from a monitoring well, the well’s reservoir is typically purged of its full volume of accumulated water (if any is present). This is done to eliminate any stagnant water in the reservoir, but Nielsen notes that extensive pumping of monitoring wells, particularly those in low hydraulic conductivity materials, may negatively bias a sample by drawing in water from both the soils in the area of interest along with water from outside the area of interest. For

monitoring bioretention practices, this could be an important consideration as a groundwater monitoring well could potentially bring in comparatively clean groundwater from beyond a bioretention practice's boundaries. This groundwater is not really representative of runoff that has been filtered through bioretention soil media and thus should be avoided. Care should be taken when purging a well to understand the potential hydraulic conductivity potential of the surrounding soils and to not purge excessive volumes of stagnant water to obtain a sample.

Additionally, the reservoir for monitoring wells must be screened to prevent clogging of the intake slits in the monitoring well pipe. Screening materials are typically a fine-grained sand, or other silica-based material like a silica-slurry, though some guidelines, notably those issued by the VT ANR, indicate that using 'clean sand or gravel' is adequate (VT ANR, 1996). The use of this material to screen the intake slits in the monitoring well pipe may also screen out pollutants artificially. Larger particulate matter would not necessarily pass through this screening material. Any pollutants bound to these particles would then be filtered out. Nielsen raises additional issues that may affect sampling of stormwater infiltrate using ground water monitoring wells such as pipe material used, agitation or aeration of sample during purging, bailing, or pumping when obtaining samples, and pressure and temperature changes which can affect water chemistry between the depth sampled and the surface (Nielsen, 2007). While potentially effective, it is clear that ground water monitoring wells are not without their limitations with respect to sampling infiltrated stormwater.

### **3.6.5** *Lysimeters – Tension (suction) and Zero-tension (pan) types*

The final potential option for obtaining samples of infiltrated water for analysis is through the use of lysimeters. Tension, or suction lysimeters are cylindrical devices that are installed in the vadose (unsaturated) zone of soil with a porous ceramic (or other material) cup leading to a reservoir. The cup is sometimes surrounded by a sand or silica slurry. A vacuum is pulled through the lysimeter's reservoir and ceramic cup, drawing water out of the pore space in soil. Water is then pumped from the lysimeter's reservoir up to the sample container for further analysis. Suction lysimeters have been used to sample various stormwater BMPs in other locations. Kwiatkowski, et al., (2007) used suction lysimeters to monitor an infiltration practice. The researchers used eight individual devices (of which three malfunctioned). The key finding from this study with respect to sampling using suction lysimeters was that timing of vacuum application is critical. The researchers used a soil moisture reflectometer to determine when runoff had infiltrated to the depth at which the suction lysimeter's intake cups were installed, in this instance about an hour. A vacuum was created and the soil sampling container of 150 ml would then fill over the course of 12-24 hours. The composite samples were then analyzed. Similar methods were used by Welker, et al. (2013) at a rain garden at Villanova University, and by Komlos & Traver (2012) monitoring an unlined infiltration-based bioretention system. Notably in the Komlos & Traver study, the suction lysimeters were installed at two depths. One was installed so that the soil pore water sample was obtained from the bottom of the bioretention's enhanced soil media at 1.2 m in depth while the second sample was obtained approximately 1.2 m below that at a depth of 2.4 m from native soils. Though the researchers do not state this in their study, this lower site was presumably a measure of the

additional treatment that might be offered by native soils underlying a bioretention practice. However, their results do not indicate that a statistically significant amount of additional treatment took place in the 1.2 m between sampling points.

Zero-tension or pan lysimeters are shallow, slightly-angled pans that are installed underneath a column of soil. Water that infiltrates through the soil column is collected in the pan where the angle of the pan causes infiltrated water to run down to a collection tube and sample reservoir. This sample reservoir can either be in an excavated structure or within a smaller pipe. The sample of infiltrated water is then obtained from the reservoir. The potential advantage that pan lysimeters have over suction lysimeters is that they only collect infiltrated water from the soil column directly above them. There is significantly less potential to collect infiltrated water drawn from the surrounding soil, thereby lessening or eliminating the possibility of inadvertently collecting comparatively more or less polluted water that could bias the sampling. The primary trade-off with pan lysimeters is that they require installation before bioretention soil media are placed or alternatively they would require the removal of existing bioretention soil media in order to place the pan, which could affect how that particular soil interacts with stormwater runoff versus how the undisturbed soil around it interacts with runoff. This makes them either less portable or more disruptive than suction lysimeters. Additionally, the introduction of an impermeable pan in the soil column has the potential to create water mounding of infiltrated runoff. This could cause water that would normally infiltrate directly downward with gravity to encounter the water mound and disperse laterally. In a review of soil vadose zone monitoring methods Fares, et al., (2009) found that zero-tension lysimeters tended to

undersample vadose soil water. This has the potential misrepresent both the quantity and quality of the water due to undersampling of the soil matrix flow.

Of the potential methods to obtain an aqueous sample from stormwater BMPs that are not necessarily designed to be monitored, suction lysimeters and groundwater monitoring wells may present the best options. The limitations are similar and include the potential to filter out particulate pollutants, the potential of certain materials used for either the sampling cup or reservoir material to alter a sample, as well as the changes potentially incurred by extracting a sample from a depth and the resulting pressure and temperature changes that may result. With respect to remoteness of monitoring sites and the need for researchers to be on-site to initiate sampling, there may be the opportunity to create automated suction lysimeters or groundwater monitoring wells, such as described by Farsad, et al. (2012). Pan (zero) tension lysimeters also show promise for this application, but may require either planning before bioretention installation to accommodate them, or disruption of an existing practice's soil media in order to put them into place.

While the results obtained from both the Hardwick and Waitsfield sites were not adequate for rigorous analysis with respect to pollutant removal, the lessons learned from the attempts to monitor them can serve other researchers or regulators with guidelines for potential methodologies that may be more successful in monitoring for pollutant removal, especially with respect to meeting potential water quality regulations.

### **3.7 ACKNOWLEDGEMENTS**

Funding for the research in Waitsfield was provided by the VT Department of Environmental Conservation's Ecosystem Restoration Program grant, obtained and



administered by the Friends of the Mad River. Design work on the bioretention practice in Waitsfield was provided by Watershed Consulting Associates, LLC and Lakeside Environmental Group. Dr. Stephanie Hurley was the primary researcher on this grant, as well as providing extensive design input on the practice, and implementing and maintaining many aspects of the practice itself.

Funding for the project in Hardwick was also provided by the VT Department of Environmental Conservation's Ecosystem Restoration Program grant, obtained and administered by the Caledonia County Natural Resources Conservation District. The practice was designed by Milone & MacBroom. The design for Cell 2 (incorporating EAF steel slag) was provided by Dr. Aleksandra Drizo of PhosphoReduc, LLC. Additionally, the Town of Hardwick provided assistance during construction, in association with Gravel Construction.

### **3.8 *Appendix 1 – Laboratory and Data Analysis Methods***

#### **Laboratory Methods**

Total Suspended Solids concentrations were determined gravimetrically by first weighing each 45 µm filter, filtering 1 L (or until filter clogging occurred), of runoff through the filter using vacuum suction and allowing the filter to dry for 24+ hours at 103° Fahrenheit, then re-weighing the filter to determine weight of sediment trapped and correcting for volume (APHA Method 2540 D).

Total Phosphorus determination was made by taking 50 ml of each sample unfiltered and refrigerated at 4° Celsius for digestion for TP. 15 ml of each sample was filtered for dissolved reactive phosphorus (DRP) analysis using a 45-micrometer glass filter and frozen for batch analysis.

TP concentrations were determined by taking 5 mL of previously-stored samples and performing a potassium persulfate digestion to convert particulate P into DRP. Samples were analyzed using the same instrument as for DRP following persulfate digestion (APHA Method 4500-P B).

Dissolved Reactive Phosphorus concentrations were determined by taking previously-filtered and frozen samples out of the freezer and allowing them to thaw completely. Analysis was conducted using a Lachat Instruments Flow Injection Analysis spectrometer to determine DRP concentrations using a standard molybdate color reaction with ascorbic acid (APHA Method 4500-P B).

Data were stored in Microsoft Excel spreadsheets and imported to JMP 12.1 statistical analysis software for computation of mean, standard deviation, and detection of outliers (if applicable). For Hardwick, data distributions were generated for all data. Additionally, matched pairs t-test analyses were run for all analytes from Hardwick. However, these analyses were run for all data in the aggregate, meaning that the pairs being matched are not, in fact, matched with respect to time. This means that the results presented from this analysis is not statistically valid.

### **3.9 *Appendix 2 – Data and Preliminary Results***

#### **3.9.1 *Village Square bioretention site – pre-construction sampling and analysis:***

Samples were taken on eight separate dates at nine separate times at the inlet to the storm drain of the Village Square parking lot as indicated in Figure 1 of the Methods section.

Table 3-1: All pre-construction sampling results from the Waitsfield bioretention site showing precipitation amounts (at time of sample, cumulative for storm, storm duration, as well as antecedent dry period), and results for TSS, TP, and DRP.

Date	Time	Precipitation	Storm	Storm	Antecedent	Prior	TSS (mg/L)	TP (mg/L)	DRP (mg/L)
		@ Sample Time (in)	Total (in)	Duration (hrs)	Dry Period (d)	Precipitation Event Total (in)			
23-Apr	9:10 AM	0.95	1.27	17	10	0.99	30.6	8.74	0.016
4-May	7:10 AM	0.33	0.34	8	3	0.42	44.76	2.56	0.020
15-May	9:00 AM	0.11	0.48	17.5	7	1.46	157.92	0.38	0.030
12-Jun	4:00 PM	0.02	0.32	11.5	3	0.28	184.63	0.468	0.065
24-Jun	8:00 PM	No Data Recorded by Weather Station			10	0.37	178.4	0.336	0.102
9-Aug	9:30 AM	0.08	0.08	9	3	0.44	23.25	0.197	0.047
9-Aug	4:50 PM	0.61	1.16	9	<1	0.08	20.29	0.133	0.054
5-Sep	8:15 AM	1.15	1.15	8.5	7	0.46	38.51	0.194	0.042

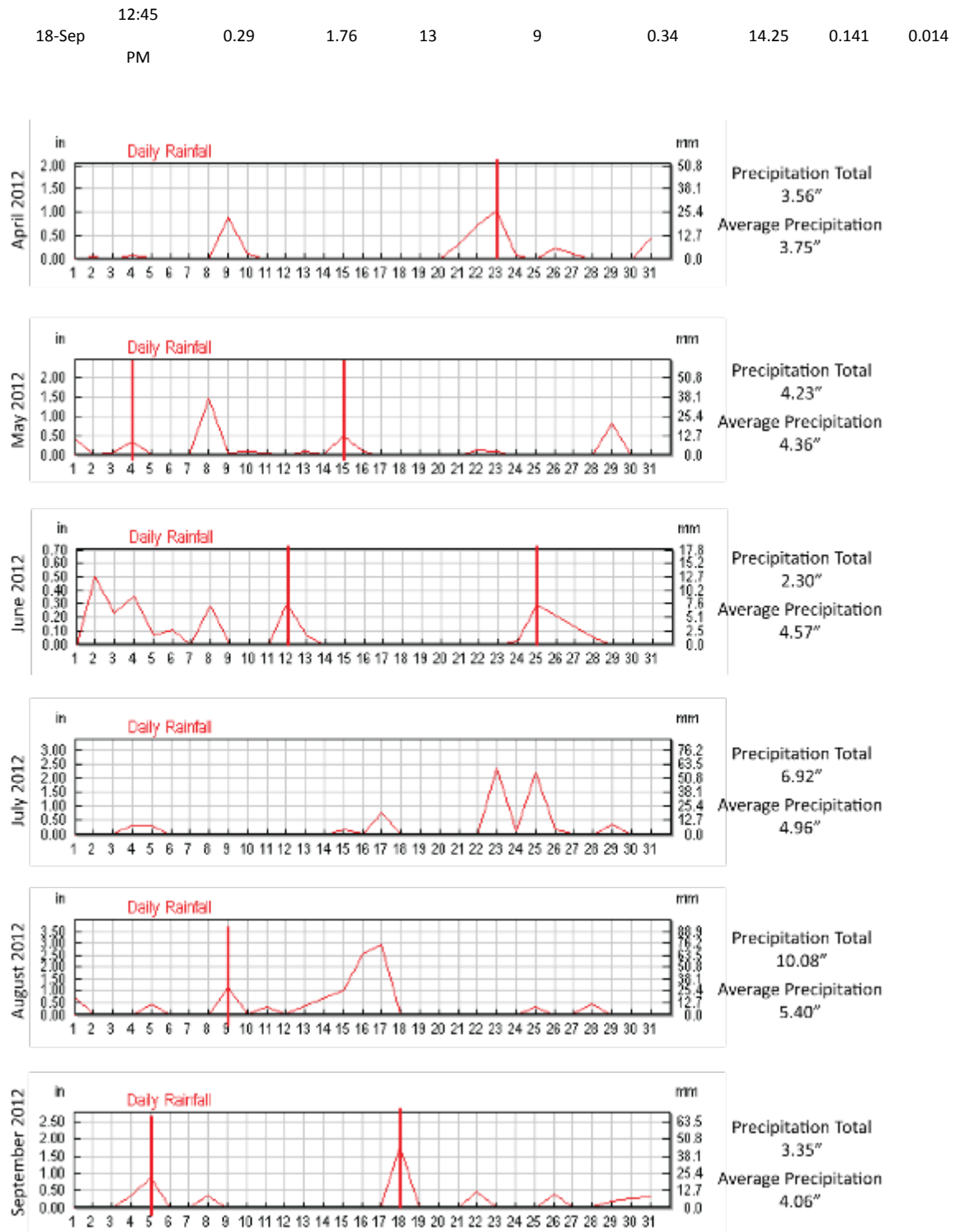


Figure 3-8: Graphs showing precipitation broken down by month for April – September, 2012 taken from a local personal weather station. Sampling events are indicated by vertical

red bars. No samples were taken in July. Note that the sample for 6-24 was for an event at 8PM that does not show up in the precipitation record for that day – the precipitation shown is for a storm beginning around 10:00PM.

### **3.9.1.1** *Total Suspended Solids – Concentrations*

Total suspended solids levels vary widely for these sample dates – the average TSS concentration is 76.96 mg/L with the standard deviation at 69.22 mg/L. Values range from a low concentration value of 14.25 mg/L to a high concentration value of 184.63 mg/L.

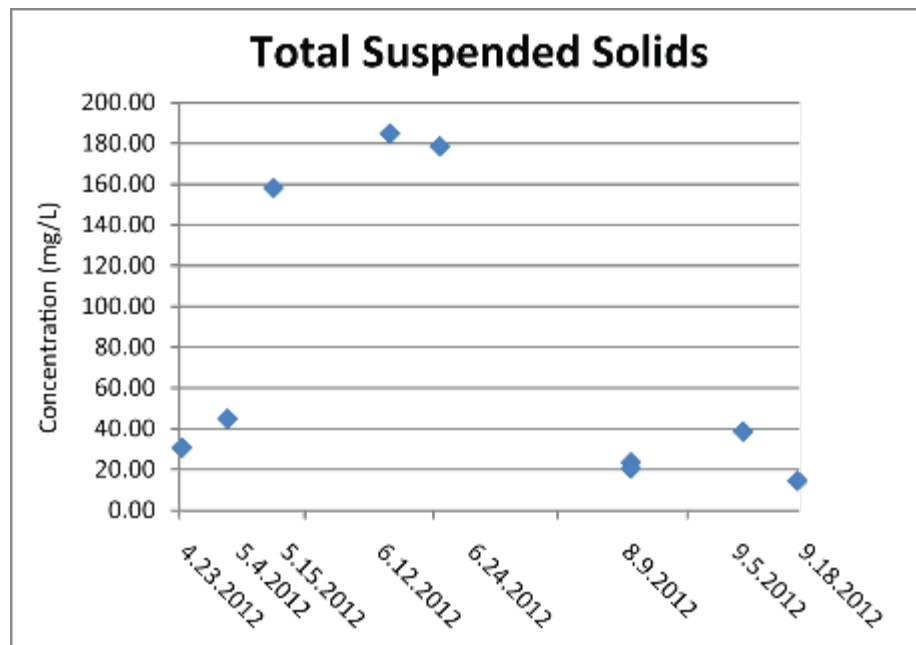


Figure 3-9: Total Suspended Solids concentrations over the pre-construction testing period. Average TSS concentration was 76.96 mg/L with a standard deviation of 69.22 mg/L.

### **3.9.1.2** *Total Phosphorus – Concentrations*

Total phosphorus concentrations average 1.46 mg/L with a standard deviation over the range of sample dates of 2.67 mg/L. This indicates the presence of outlying values that may skew the data. These outlying values are the first two samples taken on 4-23 and 5-4. If

these values are removed from the dataset, the average and standard deviation change to 0.026 mg/L and 0.12 mg/L respectively. This can be seen in the graphs below.

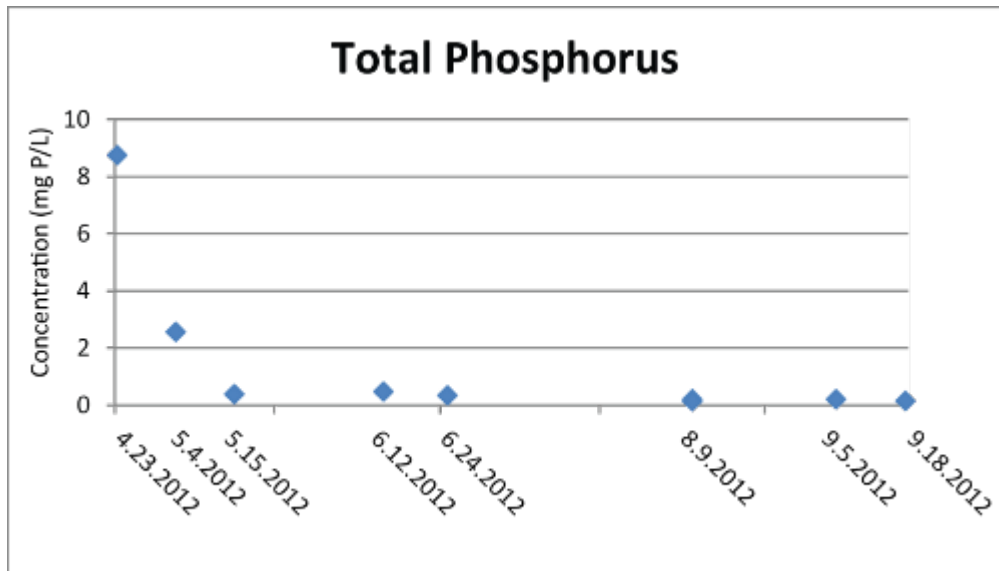


Figure 3-10: Total Phosphorus for all sampled dates – note that the sample for both 4-23 and 5-4 are above 2.0 mg/L while no other sampled event displays a concentration above 1.0 mg/L. The average for these events is 1.46 mg/L with a standard deviation of 2.67 mg/L.



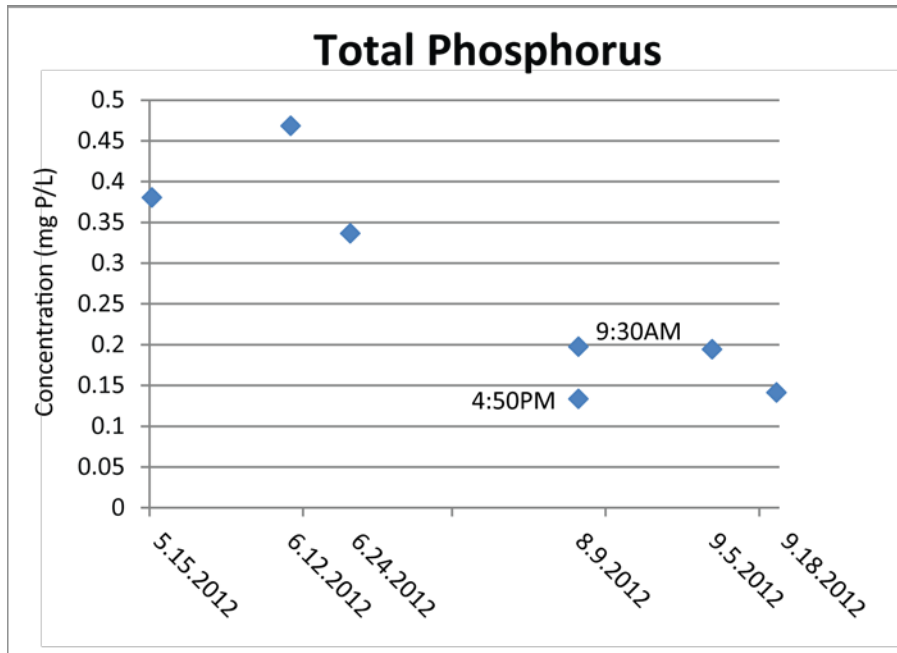


Figure 3-11: Total phosphorus for all other events exclusive of 4-23 and 5-4. The average value for these events is 0.026 mg/L (standard deviation of 0.12 mg/L).

### 3.9.1.3 Dissolved Reactive Phosphorus – Concentrations

Dissolved reactive phosphorus concentrations average 0.043 mg/L with a standard deviation of 0.027 mg/L over all sampled events. Concentrations range from 0.015 mg/L to 0.10 mg/L.

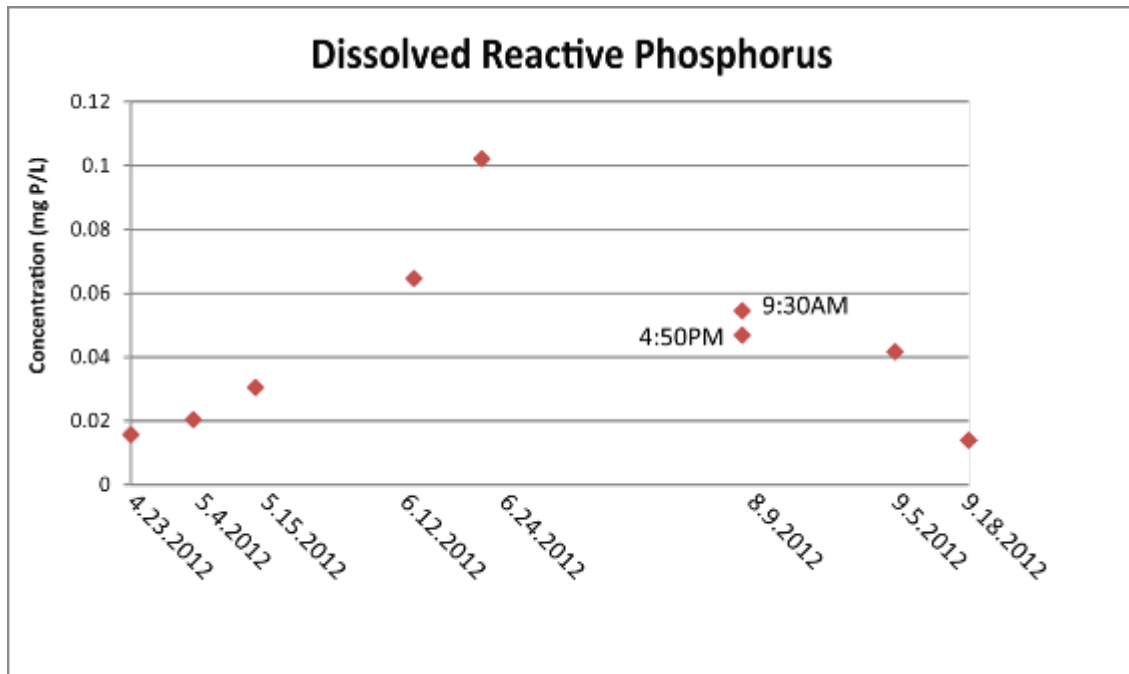


Figure 3-12: Dissolved reactive phosphorus concentrations over all sample events. Average DRP concentration was 0.043 mg/L with a standard deviation of 0.027 mg/L.

### **3.9.2** Village Square bioretention site – post-construction sampling and analysis:

The Village Square bioretention practice was constructed over the later summer and early fall of 2013. No samples were obtained after construction due to an absence of water in the sump of the perforated pipe via the sampling port. A total of five trips were made to attempt to obtain a sample. Therefore there are no results to report for this sampling port. No samples were taken from the inlet of the bioretention practice (at any one of the three RainGuardian inlets) as there was no sample to which to compare.

### **3.9.3** Hardwick Bioretention – Results

During the spring and summer of 2013, there were 38 precipitation events above 0.1". Events over 0.1" of precipitation were required for sampling as events below this threshold

did not fill and overflow the commingled forebay – no runoff water entered either cell, so no sample was possible via the perforated pipes and ISCO sampling equipment. Of these 38 events, five discrete events were sampled in some way over the course of twelve days of precipitation. These events were 5-23, 5-29 – 5-31, 6-11 – 6-13, 7-8 – 7-10, and 8-3. These events represent 3.06” of precipitation cumulatively out of the total 15.68” that fell during the period from 5-1 to 8-31, or 13% of total possible sampling events.

Of these five discrete events, influent and effluent samples were only possible in two of them: 5-29 – 5-31, and 7-8 – 7-10. The other three events had only either influent or effluent sampled. This was either due to sediment clogging of the influent intake probe, sample-enabling failure of the ISCO probes, or sample-bottle misalignment in the ISCO bottle carousel. These equipment issues were compounded by the distance from the laboratory to the field site – at approximately 1.5 hours one-way, ensuring that field personnel were on-site for the start of a precipitation event to ensure that equipment was functioning properly was problematic.

For the two events that do have both influent and effluent samples, hydrologic time-matched samples are not available. Matching the hydrograph curves is not possible in this instance – the influent pipe, while round with a known roughness and slope, lacks a weir for fine-tuning calculation of influent rates. Even if the influent hydrograph could be calculated with accuracy, neither of the two bioretention cells was lined with an impermeable barrier – runoff that entered the cells infiltrated into ground water. Therefore matching the two curves is further complicated by the loss of the majority of the water entering the system.

Finally, the two 6” perforated pipes lacked a weir at the end of the pipe, relying instead on the use of a 3” cap on the ‘outlet’ end of the pipe (the slightly lower end of the pipe near the overflow of the bioretention system). This cap, while creating the ponding necessary to sample infiltrated water, did not allow for the development of a hydrograph curve.

#### **3.9.3.1** *Analysis of All Sampled Events*

As hydrologically matching samples was not possible due to sampling issues, all sampled data was analyzed together regardless of sampled time and date.

Means and standard deviations for this data were generated by the statistical software package JMP 12.1, which was also used to conduct two sample t-tests between Influent/Cell 1 Effluent, Influent/Cell 2 Effluent, and Cell 1 Effluent/Cell 2 Effluent. Distributions were analyzed for each sample group and outliers were removed from the data using the same software package.

#### **3.9.3.2** *Total Phosphorus – Comparison of Influent/Effluent and Effluent/Effluent*

During the period of operation from 5-1 to 8-31, Influent samples had a mean of 232.80 mg/L  $\pm$  161.15 mg/L (n=55). Cell 1 TP effluent levels displayed a mean of 220.05 mg/L  $\pm$  130.02 mg/L (n = 57), whereas Cell 2 TP levels had a mean of 438.95 mg/L  $\pm$  125.48 mg/L (n = 43).

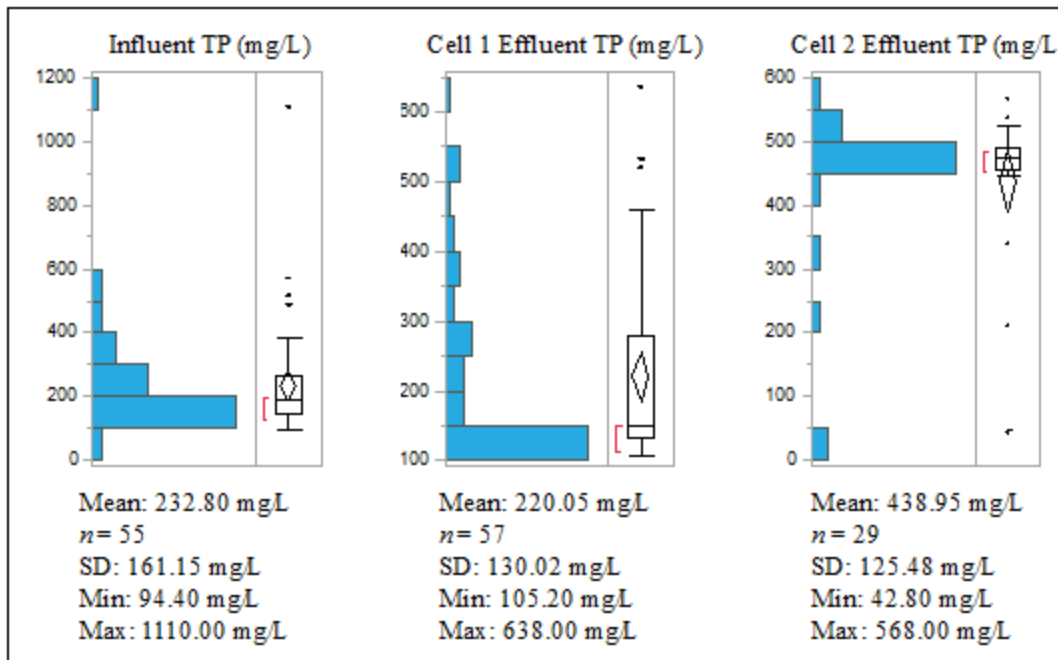


Figure 3-13: Outlier box plots for TP. Blue boxes show relative number of samples at measured concentrations. The horizontal line within the box represents the median sample value. The confidence diamond contains the mean and the upper and lower 95% of the mean. The ends of the box represent the 25th and 75th quantiles, also expressed as the 1st and 3rd quartile, respectively. The difference between the 1st and 3rd quartiles is called the interquartile range. The whiskers show the 1st quartile – 1.5\*(interquartile range) and the 3rd quartile + 1.5\*(interquartile range).

For Influent versus Cell 1 Effluent TP where  $n = 55$ ,  $t > 0.535$  with a mean difference of -17.37 mg/L (Influent TP is higher than Cell Effluent TP). The results indicate no statistically significant difference between the two sample groups.

For Influent versus Cell 2 Effluent TP where  $n = 29$ ,  $t > 0.001$  with a mean difference of 206.85 mg/L (Cell 2 Effluent is higher than Influent). The results indicate a statistically significant difference between these two groups.

For Cell 1 versus Cell 2 Effluent TP where  $n = 29$ ,  $t > 0.001$  with a mean difference of 302.28 mg/L (Cell 1 Effluent is lower than Cell 2 Effluent). The results indicate that there is a statistically significant difference between the groups.

### 3.9.3.3 Dissolved Reactive Phosphorus – Comparison of Influent/Effluent and Effluent/Effluent

A total of 74 Influent DRP samples were obtained with a mean of 31.74 mg/L  $\pm$  17.40 mg/L. 58 samples for DRP analysis were obtained from Cell 1 Effluent with a mean of 95.81  $\pm$  29.70 mg/L, while 41 samples were obtained from Cell 2 Effluent with a mean of 108.79 mg/L  $\pm$  49.74 mg/L.

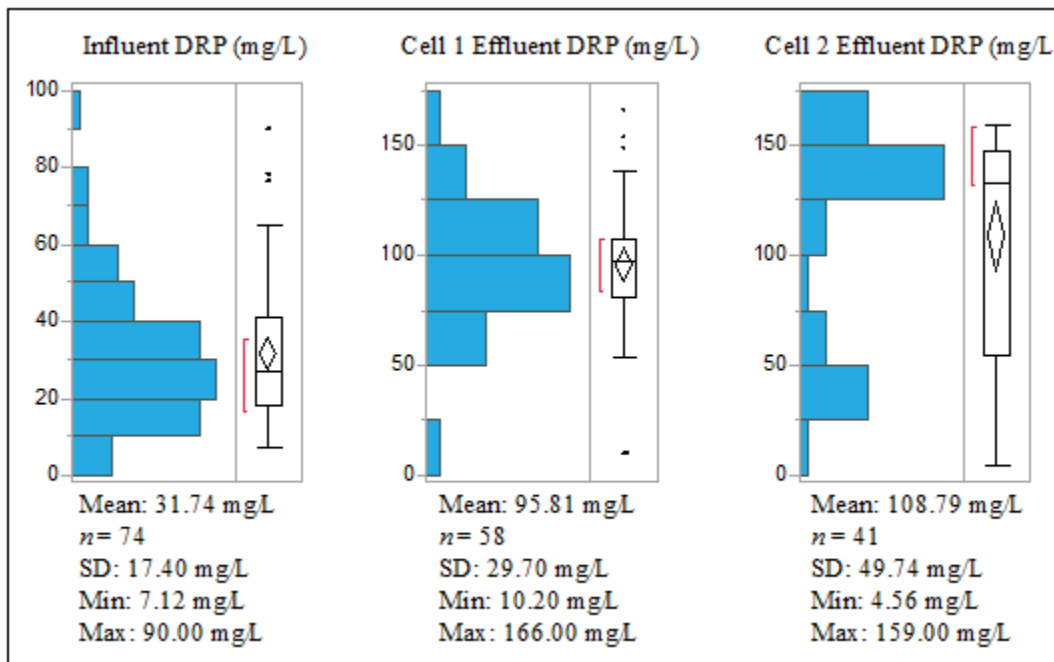


Figure 3-14: Outlier box plots for DRP. Blue boxes show relative number of samples at measured concentrations. The horizontal line within the box represents the median sample value. The confidence diamond contains the mean and the upper and lower 95% of the mean. The ends of the box represent the 25th and 75th quantiles, also expressed as the 1st and 3rd quartile, respectively. The difference between the 1st and 3rd quartiles is called the interquartile range. The whiskers show the 1st quartile – 1.5\*(interquartile range) and the 3rd quartile + 1.5\*(interquartile range).

For Influent versus Cell 1 Effluent DRP where  $n = 58$ ,  $t < 0.001$  with a mean difference of 64.36 mg/L (Cell 1 Effluent was higher than Influent). These results indicate that the two groups statistically significantly different.

For Influent versus Cell 2 Effluent DRP where  $n = 41$ ,  $t < 0.001$  with a mean difference of 72.69 mg/L (Cell 2 Effluent was higher than Influent). These results indicate that the two groups statistically significantly different.

For Cell 1 versus Cell 2 Effluent where  $n = 41$ ,  $t > 0.013$  with a mean difference of 18.21 mg/L (Cell 2 Effluent was higher than Cell 1 Effluent). These results indicate a statistically significant difference between the two groups.

**3.9.3.4** *Total Suspended Solids – Comparison of Influent/Effluent and Effluent/Effluent*

A total of 46 Influent TSS samples were obtained with a mean of 46.50 mg/L  $\pm$ 64.19 mg/L. 32 samples were obtained for TSS analysis from Cell 1 Effluent with a mean of 36.10 mg/L  $\pm$ 71.04 mg/L, while 25 samples were obtained from Cell 2 Effluent with a mean of 16.45 mg/L  $\pm$ 13.27 mg/L.

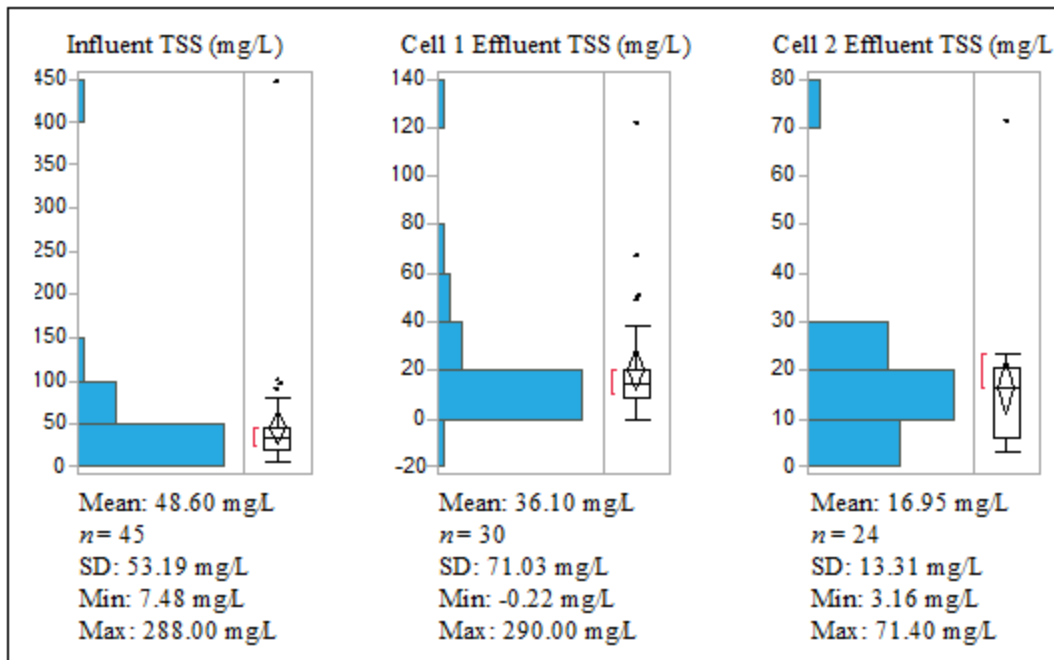


Figure 3-15: Outlier box plots for DRP. Blue boxes show relative number of samples at measured concentrations. The horizontal line within the box represents the median sample value. The confidence diamond contains the mean and the upper and lower 95% of the mean. The ends of the box represent the 25th and 75th quantiles, also expressed as the 1st and 3rd quartile, respectively. The difference between the 1st and 3rd quartiles is called the interquartile range. The whiskers show the 1st quartile – 1.5\*(interquartile range) and the 3rd quartile + 1.5\*(interquartile range).

Influent samples versus Cell 1 Effluent samples where  $n = 30$ ,  $t > 0.096$  with a mean difference of 21.88 mg/L (Influent TSS was higher than Cell 1 Effluent TSS). This analysis indicates that there is not a statistically significant difference between the two sample groups.

For Influent versus Cell 2 Effluent samples where  $n = 24$ ,  $t > 0.004$  with a mean difference of 36.55 mg/L (Influent TSS was higher than Cell 2 Effluent TSS). These values indicate that there is a statistically significant difference between the two.



Finally, for Cell 1 versus Cell 2 Effluent, where  $n = 22$ ,  $t > 0.145$  with a mean difference of 24.71 mg/L (Cell 1 TSS was higher than Cell 2 TSS). These results indicate that there is no statistically significant difference between the two groups.

Table 3-2: Summary of t-test results for all sample analytes with sample number, absolute t-value, and mean difference between sample groups. Note that these results ignore actual matching of samples and subsequently, the results are statistically significant.

Analyte	Pair	<i>n</i>	<i>t</i>	Mean Difference (mg/L)	Statistically Significant?
<b>TP</b>	Influent - Cell 1 Effluent	55	0.535	17.37	No
	Influent - Cell 2 Effluent	29	0.001	206.85	Yes
	Cell 1 - Cell 2 Effluent	29	0.001	302.28	Yes
<b>DRP</b>	Influent - Cell 1 Effluent	58	0.001	64.36	Yes
	Influent - Cell 2 Effluent	41	0.001	72.69	Yes
	Cell 1 - Cell 2 Effluent	41	0.013	18.21	Yes
<b>TSS</b>	Influent - Cell 1 Effluent	30	0.096	21.88	No
	Influent - Cell 2 Effluent	24	0.004	36.55	Yes
	Cell 1 - Cell 2 Effluent	22	0.145	24.71	No

Additionally, there is an Excel data file that accompanies this appendix titled 'Hardwick\_All\_Data.xlsx.' This data file contains all sampled runoff influent and effluent from the bioretention practice in Hardwick. There are three worksheets within the master file: 'TSS' which comprises all Total Suspended Solids data, 'P-Influent' which contains all influent sample events and the DRP and TP results, and 'P-Effluent' which contains all effluent sample events and their DRP and TP results. This file is formatted this way to accommodate printing individual pages with appropriate headings on letter size pages.

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